CHAPTER 6

Investigating Whether Dissolved Oxygen Levels in Agricultural Watercourses are Different than those in non-Agricultural Watercourses

6.1 Introduction to Dissolved Oxygen Levels in Agricultural and non-agricultural Waterways (Goal 5)

Oxygen levels in streams, rivers, and lakes play an important role in habitat availability and are a good indicator of overall water system condition (Warren et al. 1973, USEPA 1986). Although little information exists on the role of agricultural waterways in providing habitat, data suggests that over time these waterways have become inhabited with fish, including salmon species found on the NMFS's endangered species list (Berge et al. 2000, Berge 2002). Preliminary measurements have shown that un-maintained reaches may have dissolved oxygen conditions much lower than the criteria specified by the Washington State Department of Ecology (Ecology) and approved by USEPA for fish-inhabited streams (Ecology 2006). For instance, this document specifies the lowest dissolved oxygen for salmon rearing streams to be 8.0 mg/L. The premise is that prior to cleaning efforts most of the targeted waterways are colonized by a high percentage of vegetation and partially filled with anaerobic sediments. As the vegetation dies, falls into the water, and begins to decay, the bacteria decomposing the vegetation consume dissolved oxygen. This component of the dissolved oxygen balance, called the sediment oxygen demand (SOD), can effectively deplete oxygen in water systems to uninhabitable levels thereby impacting the quality of the habitat.

Currently land owners are not permitted to dredge vegetation such as Reed Canarygrass (RCG) or other invasive species from the reaches due to concerns about migrating salmon and other fish habitat. In an effort to determine the impact of maintenance activities on salmonids, the agricultural waterways in King County have been undergoing a five-year study that included a water quality component aimed at scientifically determining dissolved oxygen levels before and after dredging. This chapter of the report details the results of a comprehensive water quality study in a series of low-flow low-gradient agricultural drainage watercourses in King County, Washington. This is expected to assist King County staff and farmers in submitting their planned best management strategies based on the ability to accurately predict water quality parameters affected by these management practices.

With the assistance of KCDNRP staff, the following initial research hypotheses were posed in relation to this study component:

> Hypothesis 1: Dissolved oxygen levels in agricultural waterways do not differ from those in adjacent natural streams.

Hypothesis 2: Dissolved oxygen levels in agricultural waterways do not change as a result of maintenance activities.

These research questions were addressed through a combination of water quality monitoring and modeling. Monitoring efforts were focused on reaches that would contribute typical conditions of sites throughout the study region. Researchers sampled both agricultural and adjacent non-agricultural points for both pre-maintained and post-maintained conditions. This measurement strategy allowed researchers to draw comparisons between DO levels in maintained vs. un-maintained locations as well as note any differences between reaches flowing through agricultural land vs. those flowing through similar un-farmed areas.

Washington State University researchers tailored an existing Ecology water quality model (QUAL2Kw) (Pelletier and Chapra 2006) to fit the project's unique characteristics. This model will facilitate the County's development of management strategies for the targeted waterways. King County specifically requested dissolved oxygen predictions, therefore the results discussed in this report will be restricted to DO levels and a limited number of other significant variables which were collected along with DO data and can subsequently be used for model calibration verification.

Sediment oxygen demand (SOD) and reaeration restrictions are thought to play a large role in the degradation of oxygen levels in the King County agricultural watercourses. These man-made watercourses typically run the length of an agricultural flood plain and assist in draining the fields to a farmable saturation level. In conjunction with this study, SOD was monitored and analyzed as an important variable in understanding the DO balance and facilitate in model calibration. In addition, reaeration rates were used as a key calibration variable for the model and were evaluated for both pre and post-maintenance scenarios.

6.2 Physical System

6.2.1 Overview

The system chosen for the initial model simulation is a low-flow, low gradient agricultural watercourse network called Mullen Slough in Kent, Washington, global positioning system (GPS) coordinates (47.3609, -122.2611) and shown in Figure 6-1 as Mullen Slough C-4. The land is used primarily for flower, crop, and hay production. Portions of the land adjacent to the watercourse studies are leased or owned by Smith Brother's Dairy. The dairy applies a diluted manure mixture to the property through a manure spreader. The reach modeled was approximately 2.65 km (1.64 miles) long with a non-agriculture point-source inflow located approximately 0.8 km (0.5 miles) from the furthermost upstream (headwater) location. The non-agricultural inflow enters the valley after flowing through residential and undeveloped properties and was modeled as a point source to the main Mullen Slough reach. Data was collected at the site on September 21, 2004 before any maintenance had been performed. The average flow in the primary system was approximately 0.01 m^3/s (0.5 ft^3/s) with an average depth of 1.0 m (3.3 ft) and width of 5.0 m (16.5 ft). Average air temperature for the reach in September, 2004 was 15.4°C (60°F). Extremely dense RCG clogged most of the reach as shown in Figure 6-2.

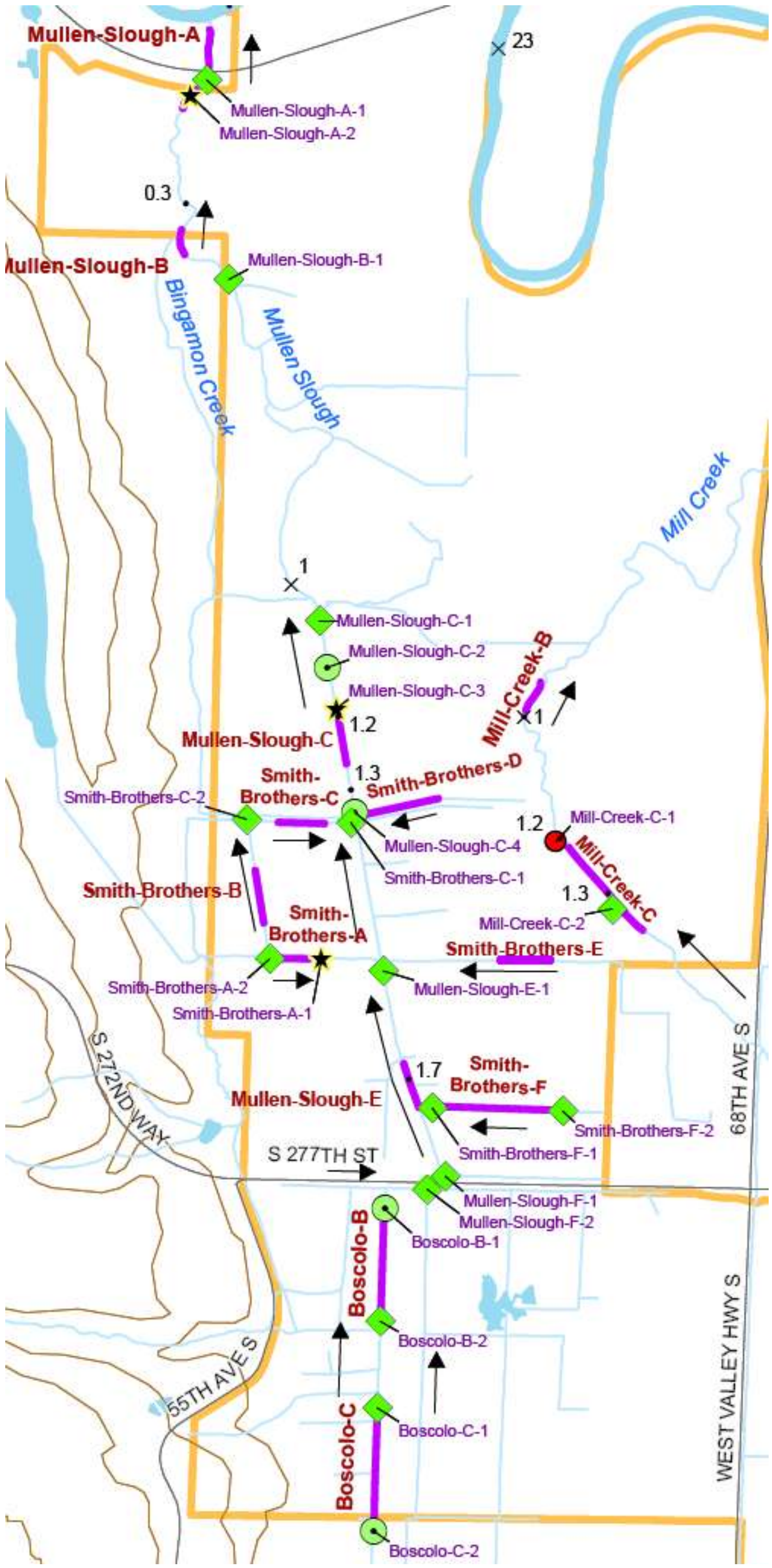

Figure 6-1. Mullen Slough study site location in Lower Green APD

Figure 6-2 Pre-dredge Mullen Slough showing dense RCG and restricted flow path

6.2.2 Dissolved Oxygen

Sites throughout the target area were selected based on their perceived contribution to the study's goals. This was accomplished by focusing on points which were either typical of agricultural or non-agricultural conditions or where maintenance operations were scheduled and subsequently performed. Samples were taken at enough points to obtain reasonable representation of reach conditions for each stage of maintenance. Samplers also recognized key inflows and outflows and took care to document corresponding upstream and downstream alterations.

Non-agricultural sites were chosen based on their proximity to their agricultural counterparts. In Kent, WA an inflow to Mullen Slough located at the Smith Brothers dairy was selected as a representative non-agricultural site (see Figure A-1). This reach, flowing from sites labeled as "Smith_Brothers_A-2" to "Smith_Brothers_A-1" in Figure 6-1, flows through a forested slope adjacent to the Smith Brother's agricultural land before joining Mullen Slough. The geomorphology of the non-agricultural reach and the main Slough differed in depth, sediment composition, shade, and flow velocity. Pre-dredged Mullen Slough was about 1.1 m (3.5 ft) deep on average while the non-agricultural inflow averaged 0.15 to 0.45 m (0.5 to 1.5 ft) in depth. The area around Smith_Brothers_1 is more forested and shaded than the man-made Slough which results in a more diverse flow path and cooler temperatures. The natural shape of the flow also provides increased aeration over natural obstacles and through pools. During the study period beavers dammed up the inflow at least once causing significant upstream flooding. Visual inspection of the beaver dam indicated that the flow exiting the beaver dam exhibited a high rate of aeration, as observed by a significant amount of turbulence, and increased velocity compared to the relatively calm upstream pond.

One of the tributaries to Mullen Slough runs in unmaintained road right of way adjacent to the Boscolo, Vilog, and Primero properties (see Figure A- **2**) which is also zoned for agriculture and is used mainly for crop and flower production. After the tributaries of Mullen Slough join together they pass under 277th . This section of Mullen Slough flows through land that is leased or owned by Smith Brother. The section this study looked at hand-cleaning their channel around summer of 2005 resulting in significant changes to the channel vegetation. The cleaning operations removed RCG and other vegetation by hand or small power tools. Plant spoils were deposited on the bank and sediment spoils allowed to settle to the channel floor. The hand dredging operations exposed most of the channel and removed essentially all vegetated shade for the reach. Flow depth after the dredge was approximately 0.3 m (1.0 ft) deep. Additionally, mechanical removal of reed canarygrass was done in August 2005.

A second non-agricultural site was chosen at the Duvall site at 124th Street (see Figure A- 3). The selected reach was Deer Creek and labeled Deer_1 for this study. Deer_1 intersects with the Pickering/Olney watercourse between Pickering_2 and Pickering_1 upstream to downstream, respectively. Deer_1 flows down a heavily shaded shallow slope out of a partially residential partially undeveloped region. It should be noted that although Deer_1 was selected as a non-agricultural comparison to the Pickering/Olney watercourse its geomorphology differs in several key areas including velocity, temperature, sediment composition, shade, and depth. Deer_1 sediment has a higher concentration of gravels and sands than the adjacent agricultural watercourse, and was visually observed to be shallower, more shaded, and swifter. The adjoining agricultural land is used for crop, flower, and hay production.

Several representative sites and their sampling locations and GPS (Global Positioning System) points are listed alphabetically in Table 6-1and Table 6-2.

Table 6-1. Duvall Sampling Sites.

Site Name	Latitude (decimal deg)	Longitude (decimal deg)	Type of Site	Date of Dredge/Clean
Deer _1	47.7117	-121.9852	non-Agricultural	NA
Olney_1	47.7158	-121.9891	Agricultural	Summer 2004
Olney_2	47.7156	-121.9864	Agricultural	Summer 2004
Olney_3	47.7141	-121.9858	Agricultural	Summer 2004
Pickering_1	47.7119	-121.9854	Agricultural	Summer 2004
Pickering_2	47.7115	-121.9855	Agricultural	Summer 2004
Pickering_3	47.7103	-121.9862	Agricultural	Summer 2004
Pickering_4	47.7091	-121.9862	Agricultural	Summer 2004

Table 6-2. Mullen Slough Sampling Sites.

Site Name	Latitude (decimal deg)	Longitude (decimal deg)	Type of Site	Date of Dredge/Clean
Boscolo_1	47.3535	-122.2588	Agricultural	Summer 2005
Boscolo_2	47.3531	-122.2600	Agricultural	Summer 2005
Boscolo_3	47.3509	-122.2601	Agricultural	Summer 2005
Boscolo_4	47.3492	-122.2602	Agricultural	Summer 2005
Boscolo_5	47.3468	-122.2602	Agricultural	Summer 2005
Mullen_1	47.3751	-122.2660	Agricultural	Summer 2005
Mullen_2	47.3713	-122.2651	Agricultural	Summer 2005
Mullen_3	47.3647	-122.2622	Agricultural	Summer 2005
Mullen_4	47.3637	-122.2620	Agricultural	Summer 2005
Mullen_5	47.3609	-122.2611	Agricultural	Summer 2005
Mullen_6	47.3578	-122.2602	Agricultural	Summer 2005
Mullen_7	47.3538	-122.2583	Agricultural	Summer 2005
Smith_Brothers_1	47.3580	-122.2635	non-Agricultural	Summer 2005
Smith_Brothers_2	47.3607	-122.2642	Agricultural	Summer 2005
Smith_Brothers_3	47.3551	-122.2587	Agricultural	Summer 2005

6.2.3 SOD

Four sites were chosen for detailed SOD studies based on their vegetation and maintenance status. The two sites selected to represent the dredged portion of the study were on Mullen Slough. Cleaning efforts removed the bulk of growing vegetation while leaving behind a large amount of sediment and decaying spoils. This sediment layer ranged in depth from approximately 0.6 to 1.5 m (2 to 5 ft) and consists of a silty material mixed thoroughly with decaying vegetation. Sediment depths were estimated based on our wading experiences during electroshocking and water quality sampling trips as well as our efforts to push a surveying rod to firm bed material. An example of the post-hand dredged waterway is shown in Figure 6-3.

Figure 6-3. Mullen Slough post hand-cleaned conditions showing thick sediment layer

A site in Enumclaw, WA owned by the Engburg/Dryer family was selected as a representative reach for un-dredged waterways. This reach called Big Spring Creek (BSC) flows through pasture land at that time inhabited by a small cattle herd and discharges to the Newaukum Creek, a tributary to the Green River. The reach is primarily inundated (see Figure 6-4) with reed canary grass with patches of clearing possibly due to livestock activity. Beneath the vegetation mat the sediment consists primarily of sand and fine gravels.

Figure 6-4. Example of Big Spring Creek vegetation

The final sampling location was on Deer Creek in Duvall (See Figure A- 3). The site was chosen as a comparison between agricultural and non-agricultural waterway sediments. The sediment characteristics closely resemble the Big Spring Creek site and consist of sandy materials. A picture of the study area is shown in Figure 6-5.

Figure 6-5. Deer Creek inflow into Pickering/Olney drainage watercourse

All four sites offer unique opportunities to observe the sediment deposition component of water quality in scenarios representative of conditions throughout the King County targeted APD's.

6.3 Methods

6.3.1 Dissolved Oxygen (DO) Data Collection

The methods used for water quality monitoring involved taking and sending samples to the WSU lab, taking stab readings with hand-held monitoring devices, and measuring diurnal fluxes using deployed logging devices (YSI Sondes).

Data at specific points along the reach were collected at both diel intervals and stab measurements over the course of three years from 2003 to 2005 with a concentration on summer months (April through September). Stab readings were collected on approximately bi-monthly schedules during that time. Diel data was collected less frequently with the majority being measured in the summer of 2004 and 2005. Diel measurements were conducted using YSI Sondes which were deployed for approximately weekly intervals and calculated dissolved oxygen, water temperature, pH, and conductivity, at 0.25-hour intervals. Grab samples were collected and sent to the Biosystems Water Quality Laboratory at Washington State University for analysis of DO, conductivity, total phosphorous (TP), total suspended solids (TSS), nitrite (NO_2^-), nitrate (NO_3^-), biological oxygen demand (BOD), and ammonia (NH_3). The four points selected on Mullen Slough included three in the primary reach and the non-agricultural inflow (Smith_Brothers_1) which QUAL2Kw treated as a point-source. At the Duvall site data was sampled at six sites including upstream and downstream from Deer Creek and within Deer Creek itself. Latitude and longitude data were determined by aerial photographs within a GIS-ArcView format. Segment geometry and lengths were calculated using standard surveying equipment.

Meteorological data was obtained for the Boeing Field weather station (located immediately north of the Kent study site) from the NOAA National Climate Data Center web site (http://cdo.ncdc.noaa.gov/ulcd/ULCD) and the University of Washington site (http://www-k12.atmos.washington.edu/k12/grayskies/nw_weather.html). The data provided includes air temperature, dewpoint temperature, wind direction and speed, cloud cover, relative humidity, solar radiation, and precipitation all measured at hourly intervals.

6.3.2 Sediment Oxygen Demand (SOD) Data Collection

SOD sampling was performed in July, 2006. Care was taken to avoid disturbing the sediments as shown in Figure 6-6. Sediment cores were removed by inserting Plexiglas cylindrical chambers into the sediment at the water interface and carefully capping the bottom of them before removal from the sediment. Figure 6-7 illustrates a typical sample prior to capping.

Figure 6-6. Sediment sampling at BSC site.

Figure 6-7. Sediment sampling chamber inserted into sediment-water interface prior to capping.

The chambers measured 15 cm tall by 9.5 cm inner diameter. They had a Plexiglas top cap and bottom base connected to threaded bolts that held both ends to the cylinder with wing nuts as shown in Figure 6-8. Seepage was in part prevented by rubber gaskets between the sediment and bottom caps.

Figure 6-8. Freshly sampled sediment cores with their caps in place.

The Mullen samples were collected in taller chambers due to limited availability of the smaller chambers. The chambers were 25 cm in height although the inside diameters were the same as the shorter ones. This discrepancy in cylinder height does not affect final SOD values due to the inclusion of water column height in the SOD calculations (see key equations in section 6.4.2.2). Sediment samples ranged from approximately 6.4 to 14.5 cm in depth with the remaining portion of the chamber being filled with stream water. This sampling technique preserves the sediment-water interface with a minimal amount of agitation. Two samples were taken at each site for the purpose of replication for a total of eight samples. The samples were kept in a cooler filled with crushed ice and transported back to the lab the day after sampling.

There was some leakage experienced in samples with more sandy sediments. This was due to the sand interfering with the chamber gaskets and preventing an adequate seal from forming. This situation was remedied by sealing the chambers externally with a line of aquarium silicone.

6.4 Data Analysis

In addition to the collected variables covered in previous sections QUAL2Kw requires specified roughness values for use in Manning's equation. Manning's roughness (n) values were calculated using software developed by the US Army Corps of Engineers called Hydrologic Engineering Centers River Analysis System (HEC-RAS) which utilized an iterative approach to obtain the roughness values which corresponded to known inputs such as flow depth, sediment depth, flow rate, and channel geomorphology (USACE 2002). Field survey data from Mullen Slough was input into the HEC-RAS model along with flow and reach characteristics for both pre and post-maintained reaches to obtain accurate results for each scenario. The model results indicated average Manning's roughness values of 0.08 for recently hand-cleaned segments and 1.9 for un-cleaned segments. These values were then used in the QUAL2Kw model for the post and pre-dredged scenarios.

6.4.1 Dissolved Oxygen (DO)

DO data collected from the sondes and stab readings were compiled and analyzed by WSU staff. This data, combined with the lab results, was organized and used to develop the QUAL2Kw model for agricultural waterways in King County.

6.4.1.1 Model Description

The model used was the Washington Department of Ecology's version of QUAL2Kw, a one-dimensional water quality model that uses Microsoft Excel as its graphical user interface and Microsoft Excel Visual Basic Application and Fortran 95 as its program languages (Pelletier and Chapra 2006). Steve Chapra and Greg Pelletier are the primary authors of QUAL2Kw, which builds on the previous USEPA model QUAL2E (Brown and Barnwell 1987). The model has been recommended by the Environmental Protection Agency (USEPA) as one of the preferred methods of predicting water quality in impaired waterways and can be found at http://www.ecy.wa.gov/programs/eap/models.html.

The model uses data from existing water systems to calculate various water quality parameters in each specified segment. The model assumes steady state hydraulics (e.g. constant inflows at all boundary conditions) and non-uniform steady flow is simulated. QUAL2Kw calculates temperature on a diurnal time scale as a function of meteorology. Organic carbon concentration is represented twofold, with fast and slow CBOD (carbonaceous biochemical oxygen demand), where fast and slow refer to the rate of oxidation for each form. Remaining non-living particulate is simulated as detritus, consisting of particulate carbon, nitrogen, and phosphorous. Oxygen levels are further simulated by QUAL2Kw's ability to calculate denitrification rates which allows the model to predict anoxia in regions where dissolved oxygen approaches or reaches zero. The interface between sediment and water is calculated within the model as a function of organic particulate settling rates, intra-sediment chemical reactions, and concentrations of overlying soluble forms. These calculated variables include SOD and nutrient fluxes. The amount of light reaching each depth in the reach is calculated based on algae concentrations, detritus, and inorganic solids. Solar radiation is calculated based on the specified time of simulation and data collection. pH is based on alkalinity and total inorganic carbon. Units for the model are metric.

6.4.1.2 Dissolved Oxygen Equations

QUAL2Kw uses both user-defined inputs and internal assumptions and defaults when calculating water quality predictions. These calculated and given values are used to produce output predictions for a specified time duration and time step. The predicted results are then paired against known data versus distance in output graphs to illustrate the degree of fitness between the two.

QUAL2Kw incorporates numerous variables and parameters to calculate a general mass-balance of constituents. Dissolved oxygen sources and sinks are calculated using the following equation:

$$\begin{aligned} S_o = {} & r_{oa}\,\mathrm{PhytoPhoto} + r_{od}\,\mathrm{BotAlgPhoto} - r_{oc}\,\mathrm{FastCOxid} - r_{oc}\,\mathrm{SlowCOxid} - r_{on}\,\mathrm{NH4Nitr} \\ & - r_{oa}\,\mathrm{PhytoResp} - r_{od}\,\mathrm{BotAlgResp} + \mathrm{OxReaer} - \mathrm{CODoxid} - \mathrm{SOD/H} \end{aligned} \tag{6.1}$$

where S_o is the dissolved oxygen source/sink term [mg O_2/L/day], PhytoPhoto is the Phytoplankton photosynthesis rate, BotAlgPhot is the bottom algal photosynthesis rates, FastCOxid is the fast reacting CBOD, SlowCOxid is the slow reacting CBOD, NH4Nitr is the ammonia nitrification rate, PhytoResp is the phytoplankton respiration rate, BotAlgResp is the bottom algae respiration rate, OxRear is the rate of oxidation reaeration, CODoxid is the COD oxidation, and SOD is the sediment oxygen demand divided by the water column depth *H*. In general, r_{xy} represents the stoichiometric coefficients for organic matter, i.e., $r_{xy} = \frac{gX}{gY}$ where gX is the mass of element X [grams] and gY is the mass of element Y [grams]. The coefficient subscripts a, d, c, and n refer to chlorophyll *a*, dry weight, carbon, and nitrogen respectively (Pelletier et al. 2005, Pelletier and Chapra 2006).

WSU researchers hypothesized that the variables with the most considerable changes between pre and post-dredged reaches would be SOD and reaeration rates. This was in part due to the limited range of accepted stoichiometric coefficients of other variables such as algal and phytoplankton growth, respiration, and mortality rates. These ranges indicated that changes within accepted parameters to any one of these rates would not result in a drastic change in simulated DO levels. COD was another consideration and therefore altered for each maintenance scenario in accordance with observed changes in collected BOD but still was not significant enough to result in the observed DO increases. The remaining unknown variables were SOD and reaeration and each were investigated in detail as a result.

SOD is an integral part of the DO equation because DO depletion occurs when oxygen-dependant microorganisms and chemical processes decay organic sediments and vegetation at the sediment-water interface in bodies of water. The ensuing processes exhaust the oxygen in the sedimentary level and subsequently become a significant DO sink in overlying water layers. The process for investigating this important parameter is covered in detail in Section 6.3.2.

Reaeration is the process by which oxygen is introduced into a water surface from the atmosphere. Most methods of measuring reaeration coefficients involve releasing a type of dye or tracer into the water and tracking its travel in units of time^{-1}. This would be difficult in pre-dredge waterways in King County due to the inability to access the water surface through thick vegetation. This is also why reaeration is hypothesized to be considerably limited in vegetated channels. A previous study involving vegetation removal and subsequent DO increases determined that reaeration and photosynthesis were the primary contributing factors (Perna and Burrows 2005). Calculating the reaeration coefficients is accomplished using equations dependant on channel morphology and flow velocity. In QUAL2Kw the reaeration rate can either be specified by the user or calculated internally by QUAL2Kw using a variety of prescribed methods. For the King County calibration, the temperature-dependant oxygen reaeration coefficient was determined to be best represented by the O'Connor-Dobbins (1958) method (a mean temperature of 20°C was assumed):

$$k_a(20) = 3.93\frac{U^{0.5}}{H^{1.5}} \tag{6.2}$$

where $k_a(20)$ is the reaeration rate at 20^{o}C [1/day], U is the flow velocity [m/s], and H is the flow depth [m].

In QUAL2Kw, flow rates can be calculated using weirs, rating curves, or Manning's equation. Manning's equation was selected for these simulations. Manning's roughness was calculated using HEC-RAS and known downstream water surface elevations from pre and post-dredge site surveys as is detailed in Chapter 13.

6.4.1.3 Calibration Method

QUAL2Kw offers two methods of calibration; by hand and an internal genetic algorithm which calibrates the model automatically based on a number of specified stoichiometric rates and constants. Due to the extensive amount of measured and known stoichiometric rates required for the auto calibration feature WSU researchers determined hand-calibration served King County better in this case. Calibration was therefore performed by hand by running the model multiple times while changing unknown parameters slightly until the model results for DO and temperature matched known values. The model plots its predictions against known values as additional points on calculated output graphs in order to facilitate this comparison. QUAL2Kw uses the root mean square error method for determining the fit of known to predicted values. This fit is calculated on the "Fitness" sheet within the model by using user-defined equations for known versus predicted differences. The known variables for the model included temperature, DO, NO_3^-, NO_2^-, pH, and conductivity. Both pre- and post- dredge conditions were calibrated and simulated. Measured input included parameters at the headwater (upstream) boundary condition as well as target data along the study reach. This information is presented in Table 6-3, Table 6-4,

Table 6-5, and Table 6-6. The input data set for post-dredge conditions was from September, 2005 and the pre-dredge data was from September 2004 except for a few variables measured in September, 2003. Both sets included data derived from lab, in-situ stab readings and diel measurements.

Table 6-3. Upstream boundary conditions used in pre-dredged calibrated QUAL2Kw model

Headwater Flow	0.014	m^3/s
Prescribed downstream boundary?	No	
Headwater Water Quality	*Units*	*12:00 AM*
Temperature	C	12.94
Conductivity	umhos	148.00
Inorganic Solids	mgD/L	76.00
Dissolved Oxygen	mg/L	0.70
CBODslow	mgO_2/L	3.08
CBODfast	mgO_2/L	1.54
Organic Nitrogen	ugN/L	15.00
NH4-Nitrogen	ugN/L	390.00
NO3-Nitrogen	ugN/L	150.00
Organic Phosphorus	ugP/L	75.00
Inorganic Phosphorus (SRP)	ugP/L	75.00
Phytoplankton	ugA/L	1.00
Detritus (POM)	mgD/L	1.00
Pathogen	cfu/100 mL	0.00
Generic constituent	user defined	8.43
Alkalinity	$mgCaCO_3/L$	74.35
pH	s.u.	6.51

Table 6-4. Measured target data for pre-dredged QUAL2Kw model

Downstream Distance (Km)	0.5	1.1	1.4	1.9
Cond (umhos)	153.00		240.00	230.00
DO (mgO_2/L)	0.70		0.40	0.22
pH	6.48		7.22	6.68
SOD-data	0.28	0.31	0.46	0.35

Table 6-5. Upstream boundary conditions used in post-dredged calibrated QUAL2Kw model

Headwater Flow	0.014	m^3/s
Prescribed downstream boundary?	No	
Headwater Water Quality	*Units*	*12:00 AM*
Temperature	C	12.94
Conductivity	umhos	148.00
Inorganic Solids	mgD/L	76.00
Dissolved Oxygen	mg/L	0.70
CBODslow	mgO_2/L	3.08
CBODfast	mgO_2/L	1.54
Organic Nitrogen	ugN/L	15.00
NH4-Nitrogen	ugN/L	390.00
NO3-Nitrogen	ugN/L	150.00
Organic Phosphorus	ugP/L	75.00
Inorganic Phosphorus (SRP)	ugP/L	75.00
Phytoplankton	ugA/L	1.00
Detritus (POM)	mgD/L	1.00
Pathogen	cfu/100 mL	0.00
Generic constituent	user defined	8.43
Alkalinity	$mgCaCO_3/L$	74.35
pH	s.u.	6.51

Table 6-6. Measured target data for post-dredged QUAL2Kw model

Downstream Distance (Km)	0.5	1.1	1.4	1.9
Cond (umhos)	152.88	151.00	165.00	176.22
DO (mgO_2/L)	2.41	3.12		4.37
CBODs (mgO_2/L)	0.65	0.57	0.96	0.87
CBODf (mgO_2/L)	0.65	0.57	0.96	0.17
NH4 (ugN/L)	310.00		150.00	
NO_3 (ugN/L)	10.00		10.00	
Alk ($mgCaCO_3/L$)	71.90	70.80		
pH	6.92	7.00	7.94	7.21
TP (ugP/L)	14760.00		2200.00	
TSS (mgD/L)	102.00		130.00	
NH_3 (ugN/L)	310.00		150.00	
SOD-data	1.94	2.20	3.21	2.67

Unless otherwise specified, model default variables were used for stoichiometric rates and were confirmed by values in the EPA rates and Kinetics manual (Bowie 1985). Several unknown parameters were estimated initially using the Rates and Kinetics Manual and then altered slightly with each model run to achieve calibration. These calibration variables include reaeration, detritus, phytoplankton concentration, algal concentration, and nitrification. The effect of vegetation removal on these rates and concentrations was difficult to measure, hence their application as unknown calibration variables instead of known inputs.

6.4.2 Sediment Oxygen Demand (SOD)

6.4.2.1 Lab set-up

Once in the lab the samples were kept in a dark 4° C refrigerator before being transferred to an intermediate cooler and then to room temperature. They were then mounted on a Phipps and Bird PB-700 Jar Tester with six rotating paddles. The experimental set-up is shown in Figure 6-9. A paddle was inserted into each chamber about 1.5 -2.0 cm above the sediment surface. The velocity of the paddles was determined using Beutel's method (Beutel 2006) for the Phipps and Bird tester paddle size:

$$V = 0.265 * RPM \quad (6.3)$$

where *V* is the velocity [cm/s] and RPM is revolution per minute.

The tester was set at a revolution of 10 RPM which equates to a velocity of 2.65 cm/s. This velocity simulated flow patterns in the chambers which has been shown to produce more stable DO measurements by homogenizing the water. This is accomplished with a velocity low enough to produce the desired results but not swift enough to cause sediment agitation.

Figure 6-9. Lab set-up with probes, paddles, and aerating tubes installed.

The samples were then aerated with aquarium pumps and tubing in darkness until they reached saturation (approximately 8.0 mg/L of oxygen). Oxygen levels were measured using DO probes inserted into the top of the chambers and sealed off. The probes were luminescent probes (see Figure 6-10) which according to Beutel et al. (2006) have an advantage over traditional membrane DO probes for their inability to absorb oxygen and their pre-calibration status. Before initiating DO measurements each chamber was topped off with deionized water and sealed with rubber stoppers to prevent any external introduction of oxygen.

A total of four measurements were run for Boscolo and BSC, five for Deer Creek, and two for Mullen Slough. Mullen Slough had fewer measurements due to limited availability of the DO probes. All samples were mounted on the Phipps and Bird PB-700 Jar Tester except for Mullen Slough because the Mullen samples were collected in chambers too tall for the apparatus.

Figure 6-10. Luminescent DO probe.

6.4.2.2 Key Equations

According to Nakamura and Heinz (1994) consumption at the sediment/water interface can be effectively modeled by assuming a two-layer condition exists consisting of an oxygen-rich upper water layer and a lower, high SOD diffusive boundary layer. For his model he also assumes a steady state system with constant horizontal velocity and smooth stream bed and walls. Previously Walker and Snodgrass (1986) represented SOD as:

$$SOD = \frac{C_{\infty}}{K_{0_2} + C_{\infty}} k_c C_{\infty} \qquad (6.4)$$

where the first term represents the biological SOD (BSOD) and the second the chemical SOD (CSOD) component. The biological term represents the biological oxidation of organic matter occurring in the aerobic portion of the sediment while CSOD refers to the oxidation of reduced compounds diffusing upwards from deeper anaerobic soils (Beutel 2006). Additionally, μ_β is the BSOD aerobic oxidation rate (g/m^2/d), C_∞ equals the bulk DO concentration in the water directly above the sediment (g/m^3), K_{02} is the half saturation constant for DO (g/m^3), and k_c is the CSOD first-order rate constant (m/d). It is worth noting that $K_{02,}$ k_c, and μ_β are also dependant on the aerobic zone depth, δ_s.

Due to his assumption that velocity was the prevailing variable in SOD concentrations, Nakamura and Heinz (1994) determined that the two parameters μ_β and k_c were not appropriate independent parameters for SOD. Further assuming that "the chemical reaction is a first-order reaction of DO concentration" the author was able to derive an equation for the volumetric rate of O_2 consumption in sediments as follows:

$$R = \mu \frac{C_w}{K_{0_2} + C_w} + k' C_w \qquad (6.5)$$

where μ is the maximum aerobic oxidation rate, k' the first order rate constant, and C_w equals the dissolved O_2 concentration at the sediment-water interface [mg/L].

For ease of differentiating between chemical and biological sediment oxygen demand Walker and Snodgrass' (1986) equation was determined to be best-suited for this study. Their model also provided the simplest method of calculation given that it is dependant on the bulk concentration of overlying dissolved oxygen which is easily measurable. Beutel further simplified this equation by setting the half-saturation coefficient (K_{02}) to zero since compared to typical SOD levels it is small, on the order of ≈0.5 g/m^3 (Beutel et al. 2006). This simplification results in the following equation:

$$SOD(C_w) = \mu + k_c \, C_w \qquad (6.6)$$

These variables are found by plotting DO as it degrades with time. SOD can then be calculated as the linear regression of DO versus time (g/m^3/d) at hourly intervals multiplied by the water column height in each chamber. These SOD values are then plotted against DO values from the midpoint of each hour interval. From this graph the remaining variables necessary can be determined, with BSOD (μ) equaling the intercept and CSOD (k_c) the slope of a linear line through the data points.

To account for different temperatures during the SOD measurements a simple equation can be used to standardize the results according to Truax et al. (1995):

$$SOD(t_1) = SOD(t_2) \cdot 1.065^{(t_1 - t_2)} \qquad (6.7)$$

For the purpose of comparison, all SOD values were standardized to 20° C in this report.

6.5 Results

6.5.1 DO Monitoring

Monitoring efforts resulted in two key findings. The first is that maintenance activities altered the DO considerably. The second is that there is a significant difference between DO levels in agricultural and non-agricultural waterways. In order to illustrate the maintenance-induced alterations plots were made of average DO levels for summer months (June through September) at the same Mullen Slough site for 2003, 2004, and 2005. The post-dredged 2005 DO data was on average 3.4 mg/l higher than the pre-dredged levels of the two prior years as is illustrated in Figure 6-11.

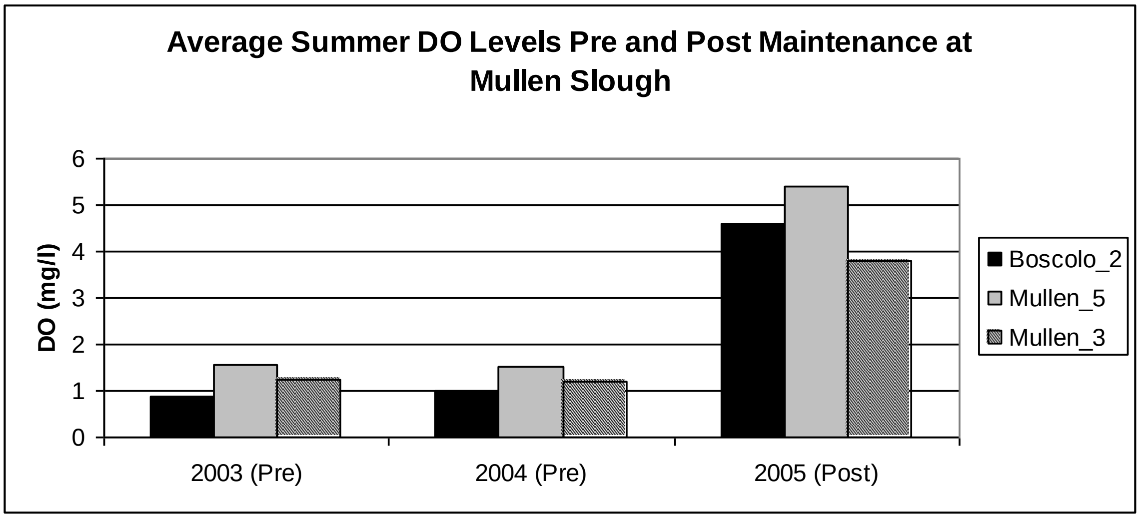

Figure 6-11. Pre- and Post- Dredge DO levels in Mullen Slough.

An observation of the diel outputs shows that DO levels in the slough are also maintaining higher levels throughout the night. Pre-dredge DO diel patterns can be seen in Figure 6-12 where the diel temperature is also shown to illustrate the relationship between the two. By contrast Figure 6-13 shows the DO levels after the bulk of the vegetation was removed upstream. Note how DO peaks around noon when the temperature is at its lowest and drops off after sunset when the temperature peaks. This inverse relationship is likely due to the increase and decrease of photosynthetic processes in the water column with a corresponding amount of solar radiation penetrating the water's surface (Wetzel 1983). Similarly, prior to vegetation removal the diel flux is much less pronounced due in part to the limited amount of photosynthesis occurring in the clogged, shaded waterway.

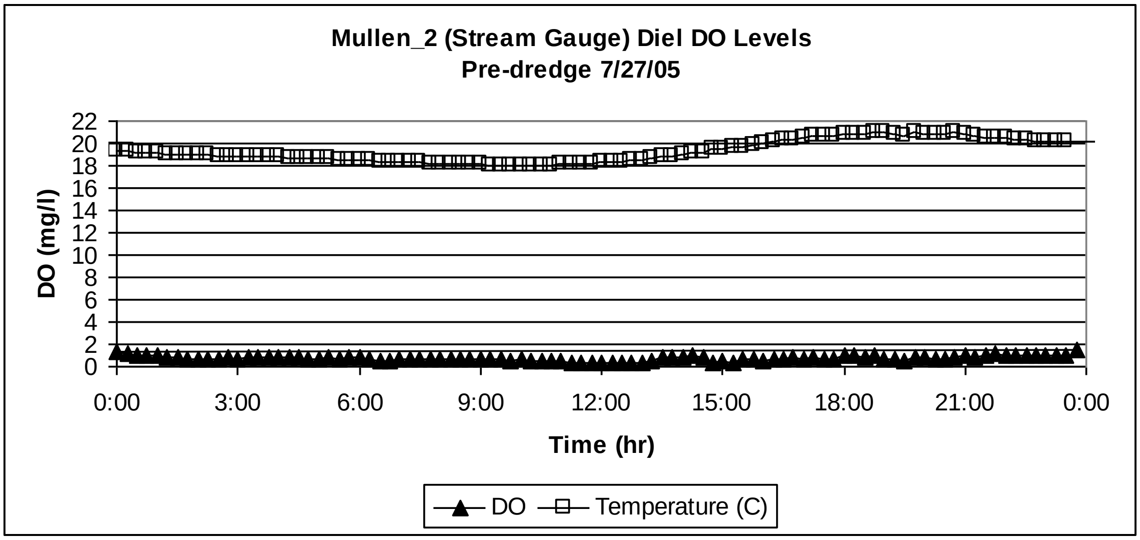

Figure 6-12. DO levels prior to upstream dredge operation's completion.

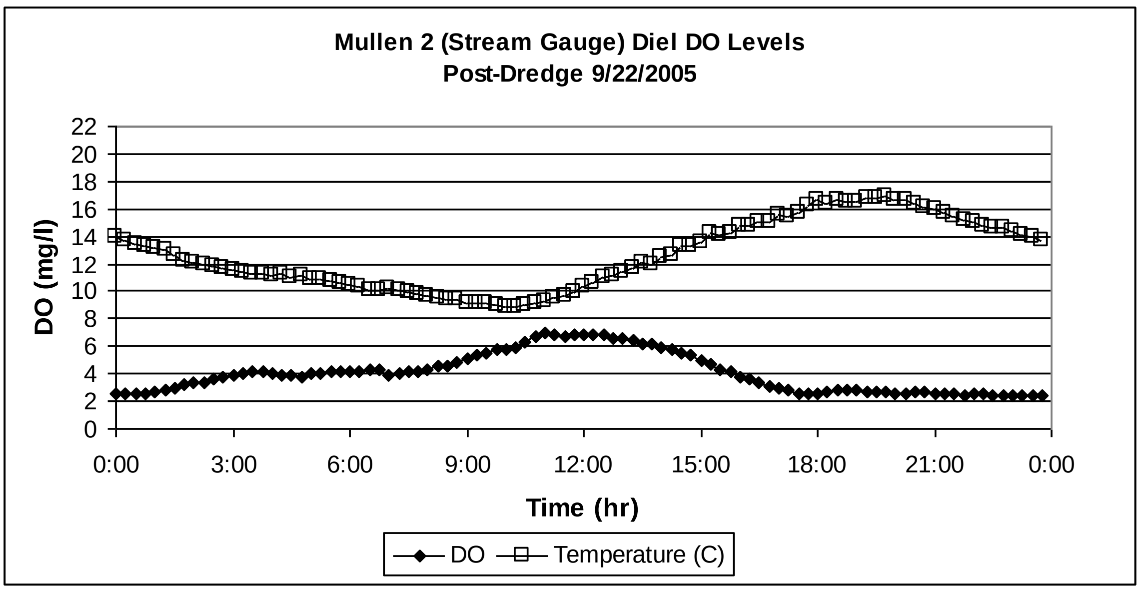

Figure 6-13. DO levels following upstream dredging operation's completion.

Additionally, there was a marked difference in measured DO levels between agricultural reaches and their non-agricultural counterparts. At the Duvall site, where sondes were deployed in the non-agricultural inflow as well as at points directly upstream and downstream, the DO showed an average increase of approximately 2.25 mg/l in the downstream portion. The diel trend at the Duvall site can be seen in Figure 6-14 which is a 24 hour sampling from September 2005. The summer stab readings (June through September) for two years at the Duvall site were averaged and graphed in Figure 6-15 to show the difference in DO levels downstream from the non-agricultural inflow (Deer_1).

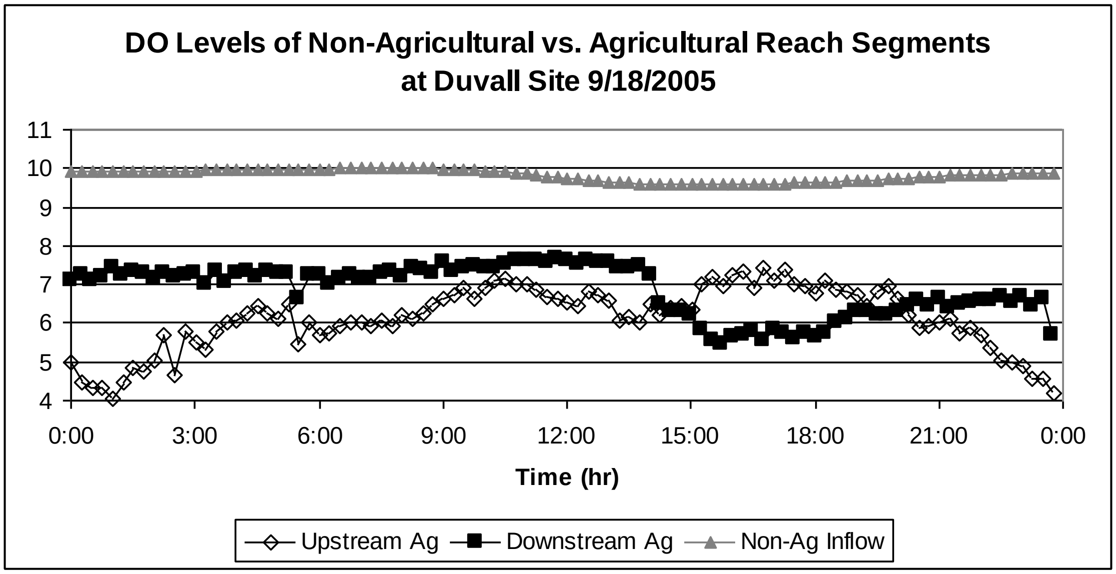

Figure 6-14. Diel DO levels upstream and downstream from a non-agricultural inflow.

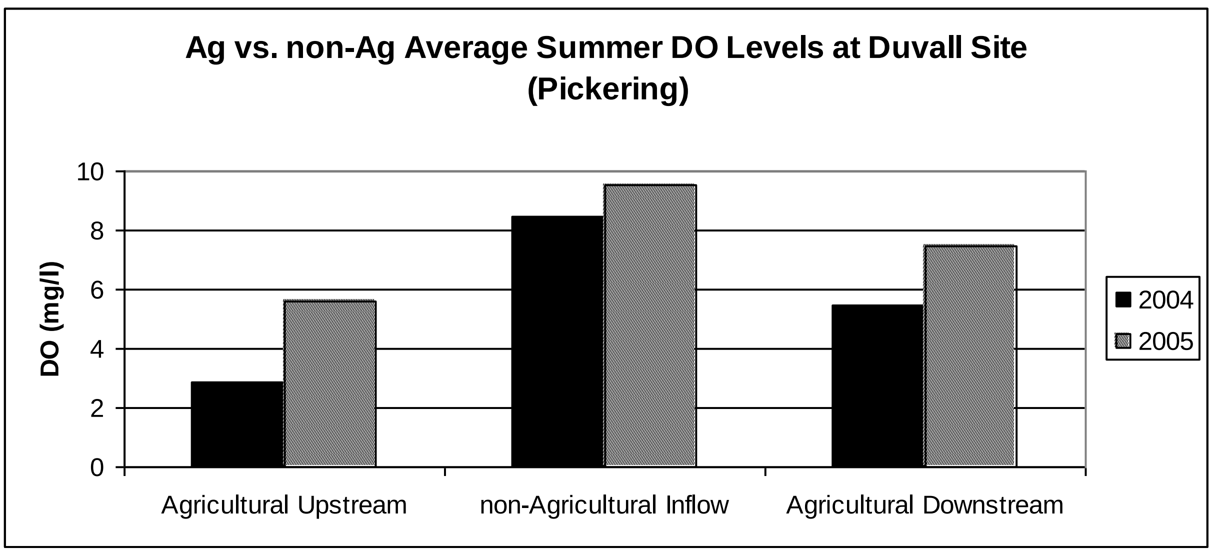

Figure 6-15. Summer stab readings at three points at Duvall site averaged to illustrate elevated DO levels in non-agricultural adjacent reaches.

The Duvall data was compiled into the graph shown in Figure 6-16 to illustrate that the elevation of DO levels in non-agricultural reaches vs. adjacent agricultural reaches was constant across all monthly stab measurements. These same values are plotted as % of DO saturation in Figure 6-17 to further illustrate this scenario. A similar trend was observed at the Mullen Slough site where the flow immediately downstream from a non-agricultural inflow (Smith_Brothers_1) rises an average of 250% (Figure 6-18). The non-agricultural inflow DO levels averaged about 5.25 mg/l higher than the adjacent agricultural segments. This trend had some variation throughout the day when the downstream section occasionally reached higher concentrations than the non-agricultural site (see Figure 6-19).

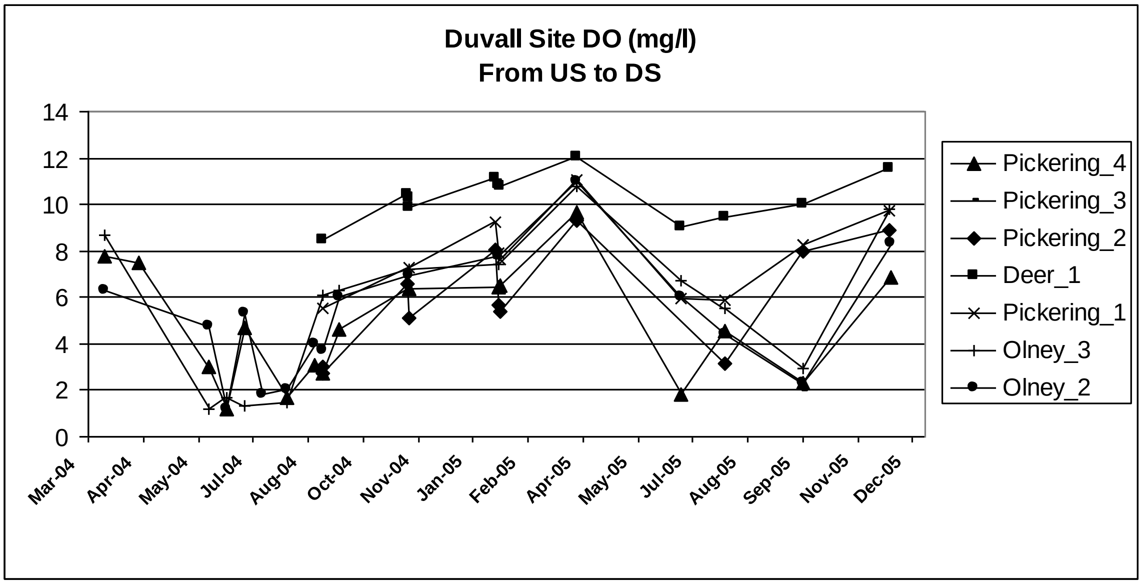

Figure 6-16. Stab DO measurements for seven Duvall sampling sites listed upstream to downstream respectively, illustrating difference between non-agricultural inflow and agricultural DO levels.

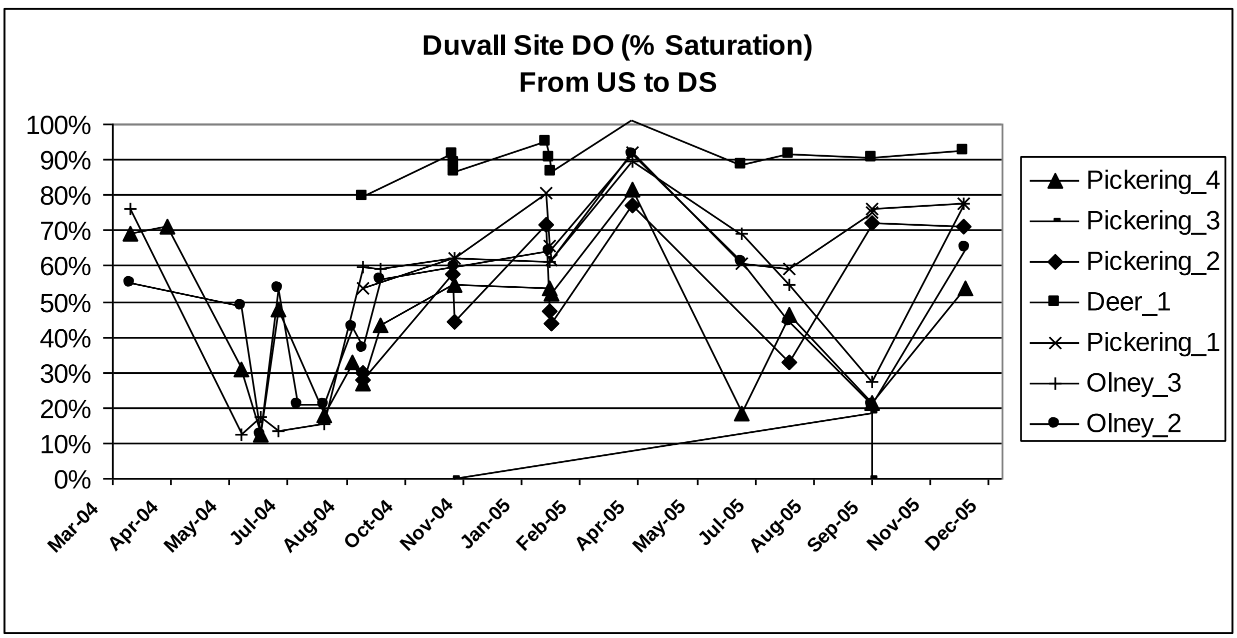

Figure 6-17. Percentage of DO Saturation of measurements for seven Duvall sampling sites listed upstream to downstream respectively, illustrating difference between non-agricultural inflow and agricultural DO levels.

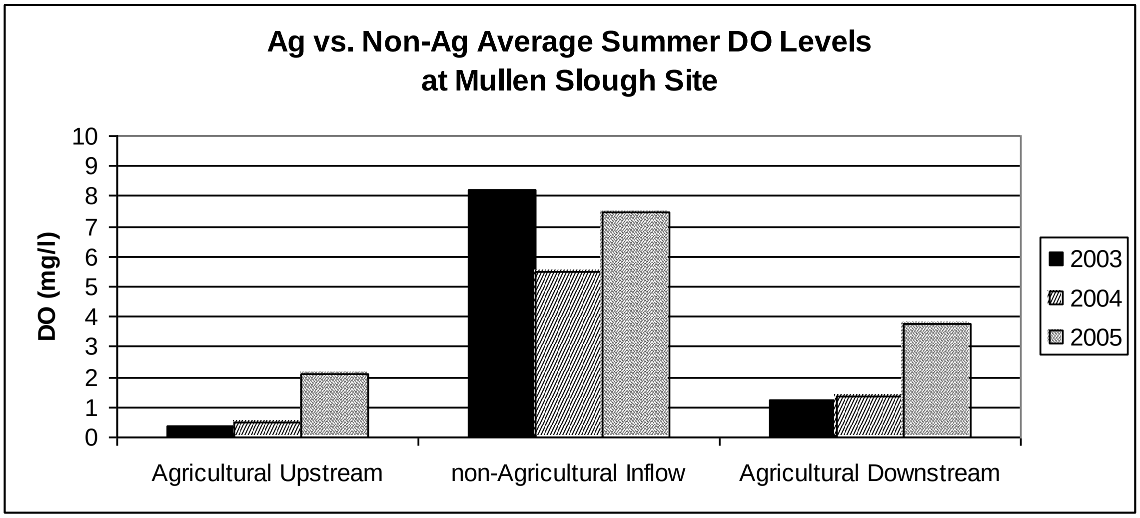

Figure 6-18. Summer stab readings at three points at Mullen Slough site averaged to illustrate elevated DO levels in non-agricultural adjacent reaches.

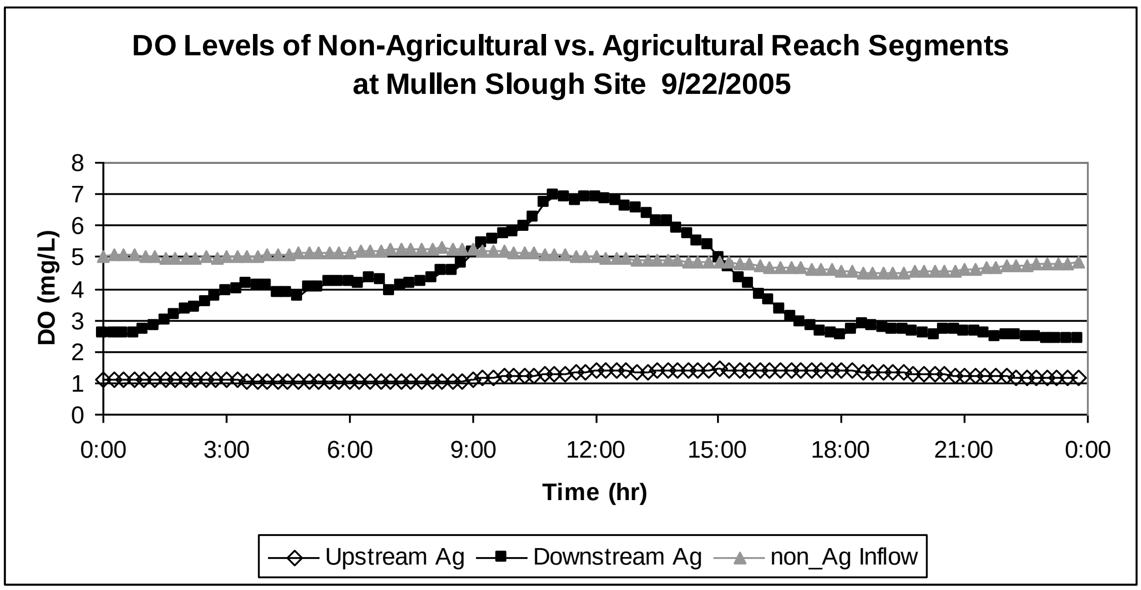

Figure 6-19. Diel plot of DO at points upstream and downstream from a non-agricultural inflow at the Mullen Slough site.

6.5.2 Model Calibration Results

In order to represent pre- and post- maintenance conditions, two scenarios were calibrated in QUAL2Kw. Calibration of the pre-dredged condition resulted in a total root mean squared error of 6.0 calculated for four variables (temperature, DO, pH, and conductivity) with each having an equal weight factor of 1. The individual temperature value was 1/.03 or 33.3 and the DO error was 1/.15 or 6.7. The output graph for simulated vs. known DO data from the calibrated run is shown in Figure 6-20. Calibration of the post-dredged condition produced a total root mean squared error of 5.0 for the same four variables as the pre-dredged scenario. Its individual temperature and DO errors were 1/.25 or 4.0 and 1/.38 or 2.6 respectively. The fit of predicted to known DO data for post-dredge calibration is illustrated in Figure 6-21. QUAL2Kw predicted post-dredge DO values to be approximately 4.0 mg/l which was a 2.2 mg/l (over 120 %) increase over the pre-dredge predictions.

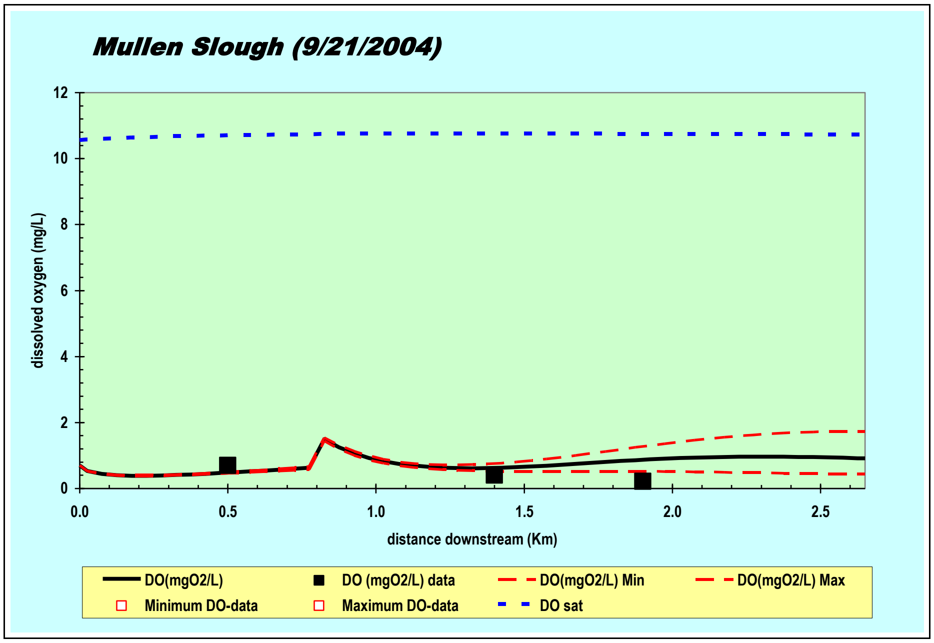

Figure 6-20. QUAL2Kw output graph of DO for calibrated pre-dredged conditions. Lines represent simulated results while blocks represent known values.

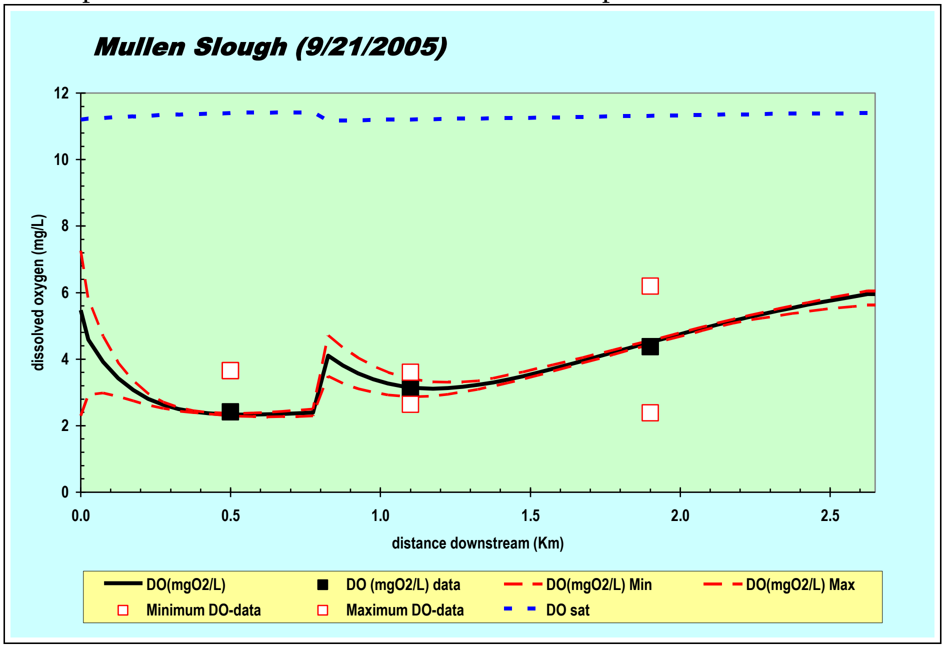

Figure 6-21. QUAL2Kw output graph of DO for calibrated post-dredged conditions. Lines represent simulated results while blocks represent known values.

Sensitivity analyses on SOD and reaeration rates were performed and the model's corresponding errors evaluated. SOD and reaeration were assumed to be the two most sensitive parameters because of their relative uncertainty compared to other model inputs. The analysis was performed on the post-dredged calibrated model with only the specified parameter being altered by values of ten and fifty percent. The fitness values were then recorded for both temperature and DO, the two target parameters of the study. The results were tabulated in Table 6-7 and demonstrate similar patterns between reaeration rate alteration and that of the SOD. Note that the final calibration resulted in the fitness values in the last row of the table.

Table 6-7. Calibration fitness outputs from using the root mean squared error method for temperature and DO QUAL2Kw predicted values vs. known inputs with changes in reaeration and SOD input values of ± 10 and 50 %, respectively.

	Reaeration		**SOD**	
Variation	**Temperature Fitness**	**DO Fitness**	**Temperature Fitness**	**DO Fitness**
+10%	4.0	2.2	4.0	2.9
-10%	4.0	2.9	4.0	2.3
+50%	4.0	2.3	4.0	2.7
-50%	4.0	0.7	4.0	1.2
calibrated	**4.0**	**2.6**	**4.0**	**2.6**

When the reaeration rate was increased by 10 percent, the DO fitness decreased by approximately 15 percent. When the reaeration rate was decreased by 10 percent, the DO fitness increased by approximately 11 percent. An alteration in the reaeration rate of plus or minus fifty percent caused a DO fitness decrease of 11 percent and 73 percent, respectively. In contrast, the temperature fitness did not change at all which indicates DO is a much more sensitive parameter to model than temperature. For SOD deviation a change of plus or minus ten percent caused a DO fitness increase and decrease of 11 percent. An increase or decrease of 50 percent resulted in a DO fitness increase of 4 percent and decrease of 54 percent, respectively. Again, by altering the SOD inputs temperature maintained a consistent fitness of 4.0. The changes in inputs that resulted in increased fitness values do not necessarily represent a better overall fit. The root mean square error method removes any limitations caused by some variables being above or below the accepted mean and instead combines any deviation into one squared value. A combination of visual inspections of predicted versus known values and the fitness method was employed to obtain the best calibration possible.

A second analysis of QUAL2Kw was performed to determine how the model's temperature predictions compared to the Heat Source model submitted to King County in conjunction with the study. Both diel and longitudinal results were plotted on the same axis for both models to facilitate this objective. Both demonstrated acceptable fitness between the models as is shown in Figure 6-22 and Figure 6-23.

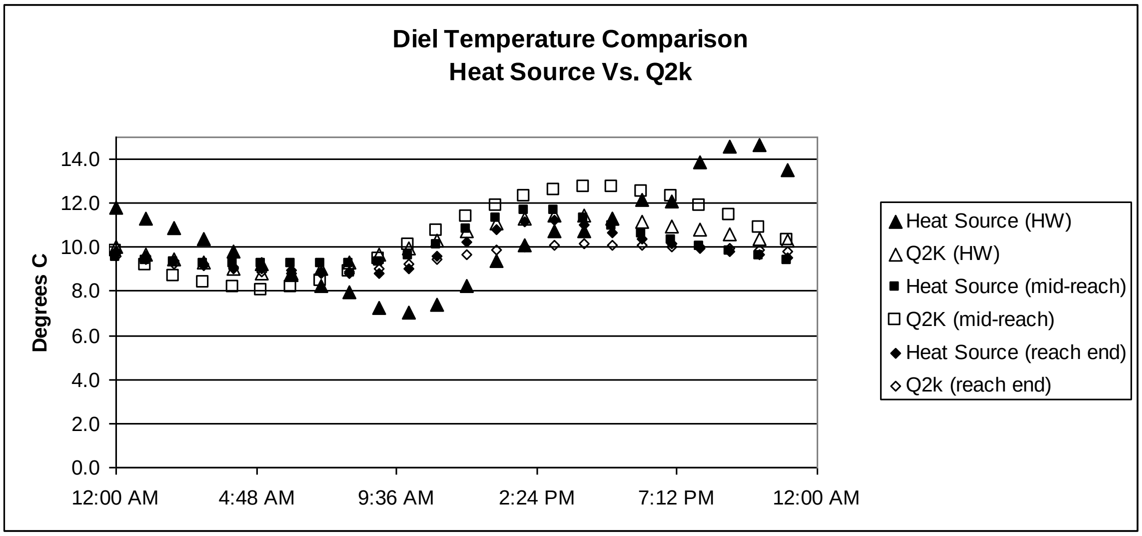

Figure 6-22. Predicted diel temperature comparison between Heat Source model and QUAL2Kw.

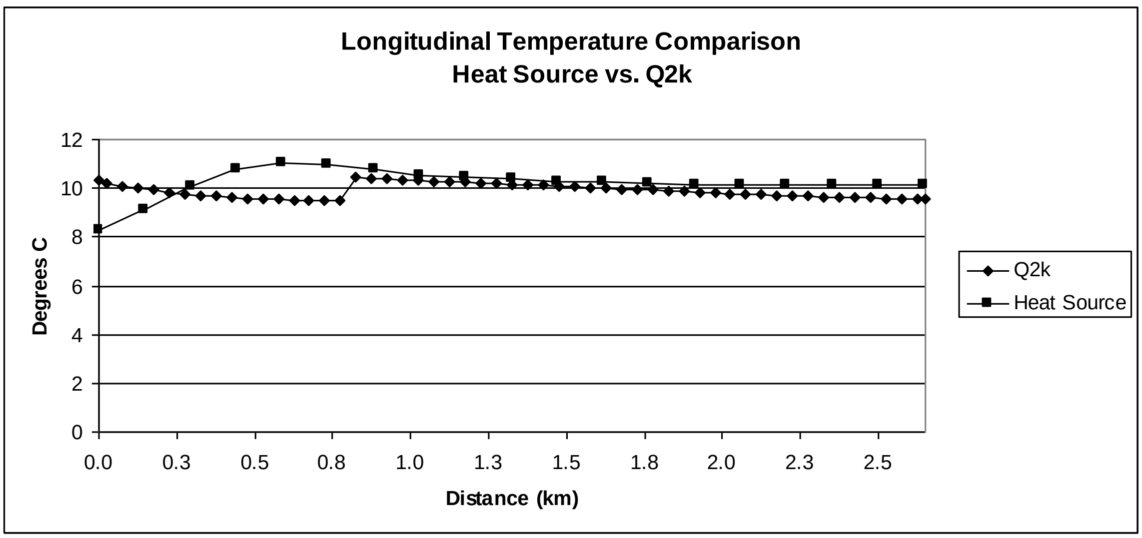

Figure 6-23. Predicted longitudinal temperature comparison between Heat Source model and QUAL2Kw.

6.5.3 Sediment Oxygen Demand (SOD)

An evaluation of the sediment cores' moisture content resulted in Table 6-8. Dredged sites Mullen and Boscolo had the lowest moisture content while those undredged or non-agricultural had moistures of approximately 28% higher. These samples were taken from the top 1.5 inches (3.8 cm) of the sediment/water interface from the cylinders used in the SOD calculations.

Table 6-8. Average moisture content of sediment cores taken from four sites.

Site	Average Moisture Content
BSC	62.6%
Mullen	31.5%
Deer Creek	51.4%
Boscolo	26.7%

The time it took for the dissolved oxygen to reach zero from saturation levels ranged from approximately 3.5 hours to 35 hours. Figure 6-24 illustrates the process of DO degradation over time. A difference in time required to degrade the DO was observed between the runs. The last run in particular demonstrated a longer duration to reach zero from saturation than the other three runs. This could be due to the fact that over time as the samples were exposed to room temperatures the mortality rate of microorganisms responsible for oxygen consumption at the sediment/water interface began to increase. This decrease in oxygen consumption would in turn cause the overall time of complete degradation to extend. Figure 6-25 shows a representative plot of calculated sediment oxygen demand at hourly intervals versus dissolved oxygen at the midpoint of the hour segments. From this and similar plots, the BSOD and CSOD values were calculated using the intercept and slope of best-fit linear lines. These lines were limited to between 1.0 and 7.0 g/m^3 DO due to increased scatter below and above those values. In some cases values slightly higher or lower were included where a limitation of measurement points required it.

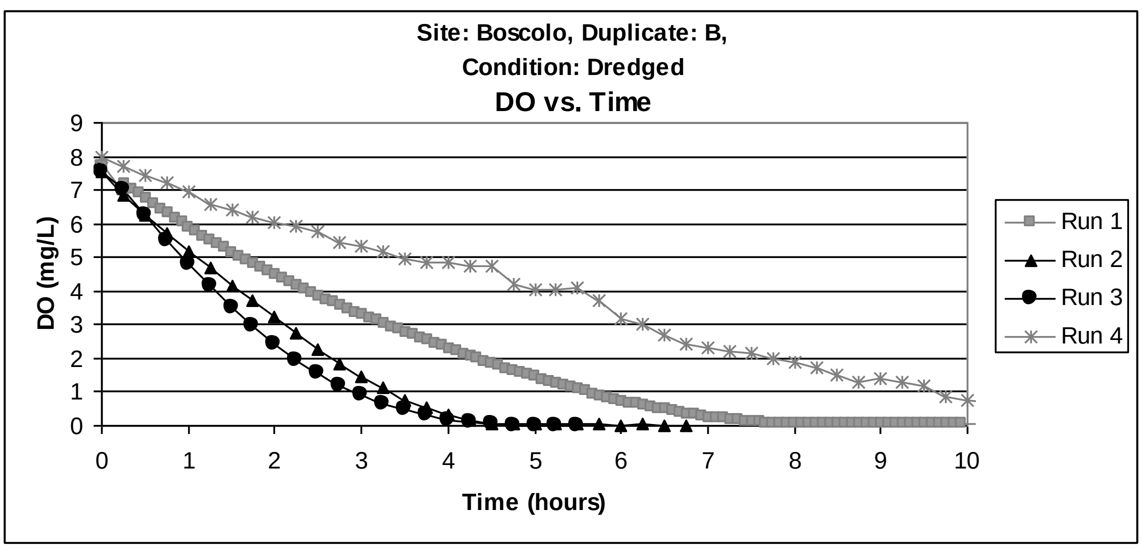

Figure 6-24. DO decreasing with time in a sealed SOD chamber with sediment from a dredged site, SOD measurements were performed four times.

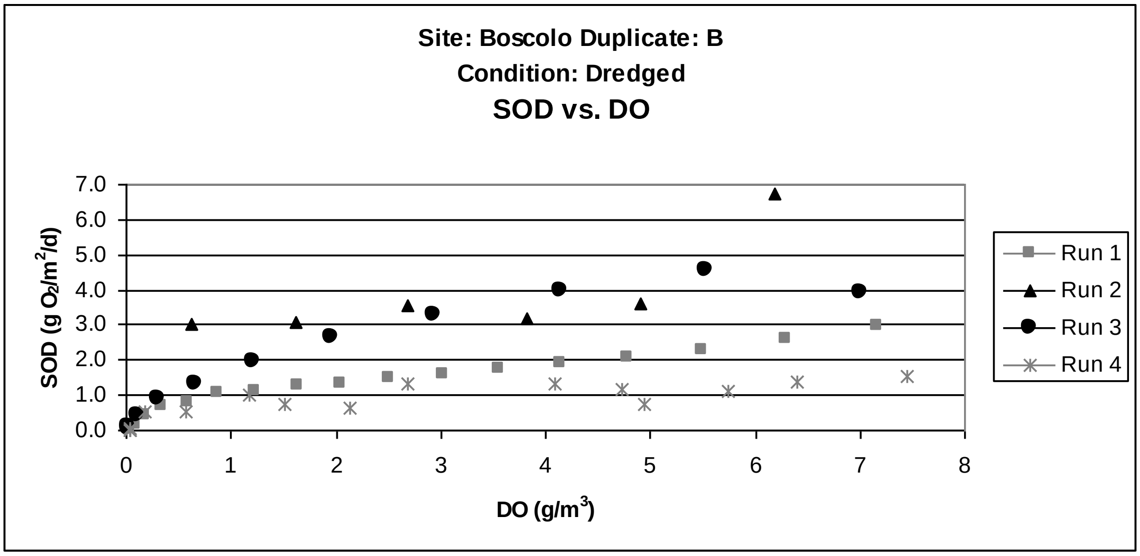

Figure 6-25. Total Sediment oxygen demand at a dredged site plotted against hourly DO intervals.

Total SOD_5 values at 20°C differed for each site with the largest value at the hand-cleaned Boscolo site (2.90 $g/m^2/d$) and the smallest occurring at the un-maintained BSC site (0.45 $g/m^2/d$). The SOD_5 results for all four sites (each tested twice) are shown in Figure 6-26.

The chemical component of the SOD (CBOD) dominated the process for each site with values ranging from 0.41 to 2.21 $g/m^2/d$. In contrast, the biological component (BSOD) ranged from 0.26 to 1.62 g $O_2/m^2/d$. Figure 6-27 illustrates the collective contributions of BSOD and CSOD to the total SOD values.

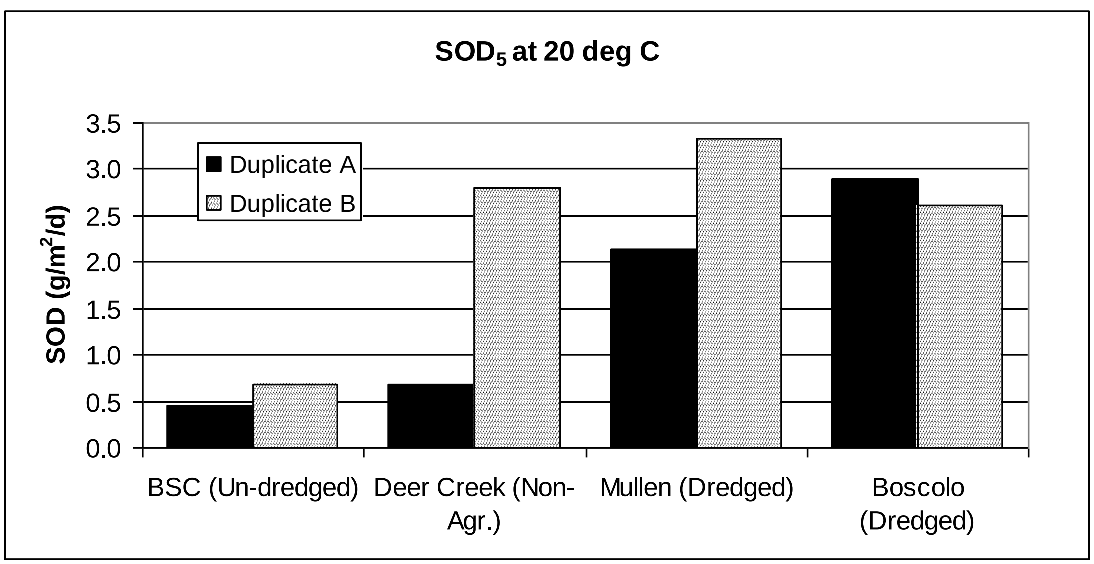

Figure 6-26. Averaged total SOD at a DO of 5 mg/l for four sites with two duplicates per site.

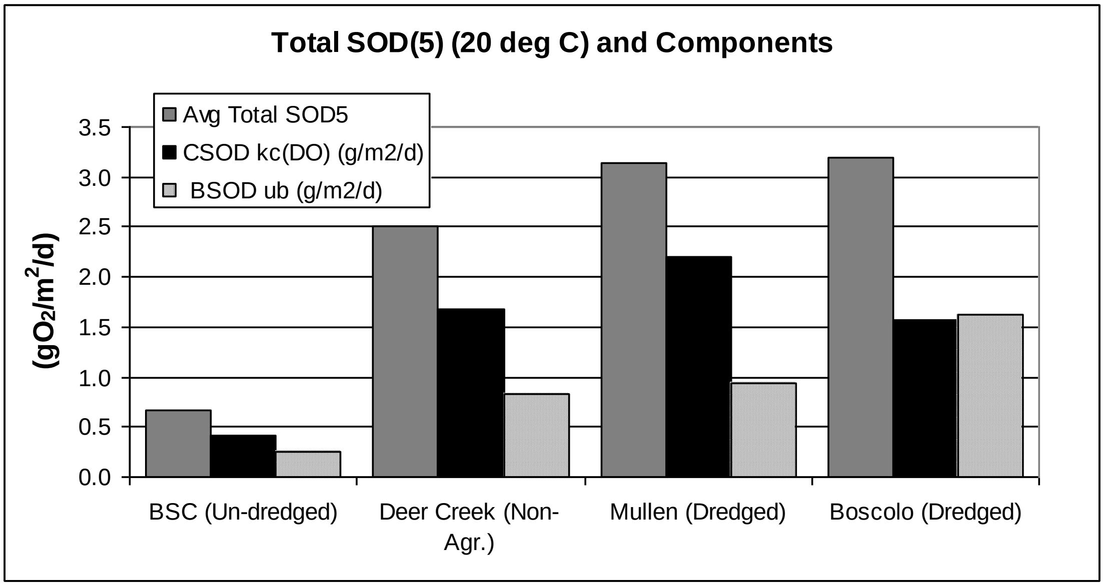

Figure 6-27. Individual contributions of biological and chemical sediment oxygen demand processes to total SOD.

R^2 values were taken across the data sets used to derive SOD values (like those in Figure 6-25) to analyze scatter and averaged from 0.16 to 0.97. SOD values were calculated for three different DO values, 2, 5, and 10 mg/l using the chemical and biological coefficients found by using methods described in Section 6.4.2.2. See Table 6-9 for individual calculated values and Figure 6-28 for the graphed results for each sampling site.

Table 6-9. Summary of lab measurement results for four sites with two sets of duplicates each run two to four times.

Site	Dup	Date	K_c (m/d)	μ_b ($g/m^2/d$)	R^2	SOD_2 (20°) ($g/m^2/d$)	SOD_5 (20°) ($g/m^2/d$)	SOD_{10} (20°) ($g/m^2/d$)
BSC (Undredged)	**A**	**Average**	**0.08**	**0.15**	**0.97**	**0.26**	**0.45**	**0.77**
		Std Dev	0.03	0.20		0.13	0.06	0.09
	B	**Average**	**0.09**	**0.37**	**0.90**	**0.46**	**0.67**	**1.04**
		Std Dev	0.04	0.13		0.14	0.23	0.40
Deer Creek (Non-Agricultural)	**A**	**Average**	**0.20**	**0.50**	**0.55**	**0.74**	**1.23**	**2.05**
		Std Dev	0.14	0.16		0.32	0.65	1.20
	B	**Average**	**0.31**	**1.82**	**0.72**	**1.98**	**2.74**	**3.99**
		Std Dev	0.26	1.15		1.24	1.78	2.76
Mullen Slough (Recently Dredged)	**A**	**Average**	**0.24**	**1.27**	**0.16**	**1.51**	**2.13**	**3.16**
		Std Dev	0.01	0.14		0.08	0.04	0.02
	B	**9.07**	**0.65**	**0.61**	**0.21**	**1.65**	**3.33**	**6.13**
Boscolo (Recently Dredged)	**A**	**Average**	**0.27**	**2.03**	**0.78**	**2.21**	**2.90**	**4.06**
		Std Dev	0.11	0.38		0.45	0.70	1.15
	B	**Average**	**0.36**	**1.22**	**0.68**	**1.68**	**2.62**	**4.17**
		Std Dev	0.24	0.63		0.85	1.40	2.39

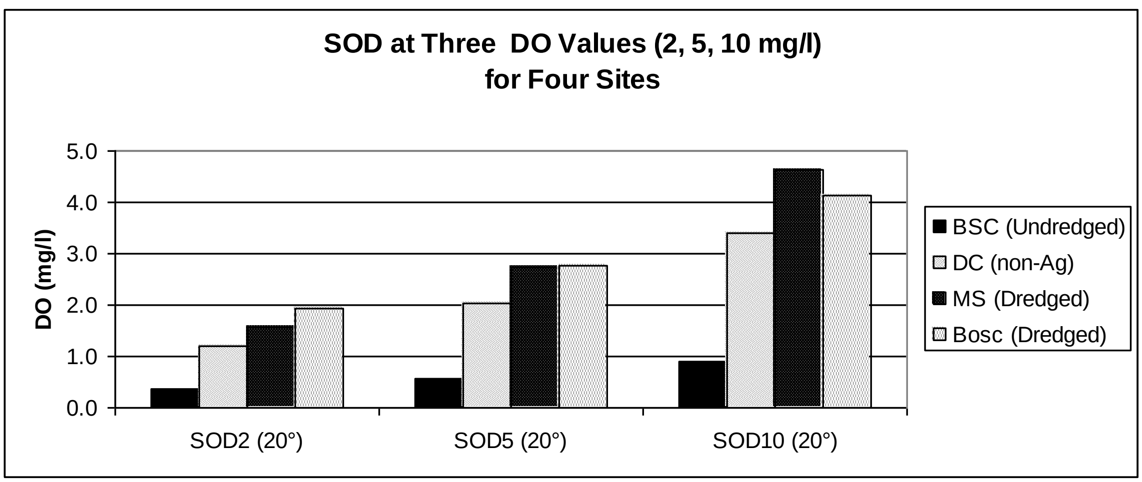

Figure 6-28. SOD at selected DO values (2, 5, 10, mg/l) for each site.

6.6 Discussion/Conclusions

6.6.1 SOD Accuracy

R^2 values for SOD versus DO plots varied from 0.16 to 0.97. This value demonstrates the amount of scatter occurring during testing. The lowest R^2 values were in the unmixed Mullen Slough cylinders, however their averaged overall SOD agreement was the best of the study. The high degree of scatter found on the Mullen plots follows a developed model which demonstrates that even the slightest mixing during measurement results in a much more linear plot due to a more uniform oxygen level crossing the DO probes (Beutel et al. 2006). In order to avoid low R^2 values and obtain more symmetrical plots, a minimum mixing velocity apparatus is typically recommended for SOD measurements of this method.

Standard deviations of SOD_5 across the runs for each site ranged from .04 to 1.78 with an average of 0.69 g/m^2/d. This was in part due to differing slopes between the first and last measurement runs. Several samples demonstrated a similar trend to take longer durations to consume the oxygen than for previous or subsequent runs. This could be due to variations in the mechanical stirring device velocity or the introduction of small amounts of oxygen into the system between runs. Related studies resulted in similar differences of up to 20 hours between runs (Beutel 2003). The longest difference in DO consumption for this study was 20 hours which was within acceptable ranges.

6.6.2 SOD Results

Pre-dredged SOD values were lower than post-maintained channels. These results were consistent across the replicates and duration of the study which gives a degree of confidence in the data and measurement procedure. The elevated SOD values in hand-cleaned waterways are possibly due to the amount of loose organic sediments and decaying vegetation remaining after the maintenance operations. The characteristics of the sediments indicate this since the two sites consisting of sandy sediments demonstrated the lowest SOD values. The highest values were found at locations where a fine sediment dominated. Age of sediments may also play a part in the SOD differences, however it was not possible to determine the duration sediments had been at each site. The microorganisms in oxygen-limited sediments such as those at the undredged sites could have been repressed until oxygen was re-introduced in the lab reaeration process. In contrast, the microorganisms in recently dredged systems had a fresh influx of oxygen and nutrients possibly increasing their activity and demand for oxygen. Since SOD is dependant on the amount of oxygen available at the sediment-water interface, an increase in resuspension results in increased sediment available for consuming oxygen. This was demonstrated by the SOD measurements for the undredged site which had more mature sediments and much less organic material in suspension. In contrast, the hand-cleaning operations left a large quantity of easily agitated material in the streambed which presents a significant DO sink via SOD processes. The channels have sandy firm beds under the layer of silt and it could be argued that if the channels were dredged down to this original sand they would demonstrate SOD values that would enhance already improved dissolved oxygen levels. To prove this theory sediment cores should be sampled from mechanically-dredged locations and run through the SOD measurement procedure and compared to hand-cleaned ones such as Mullen and Boscolo. Mineralogy studies examining sediment composition would also assist in quantifying differences.

6.6.3 Case Study

In order to investigate whether complete vegetation removal would increase DO levels even further than hand-removal techniques, simulations of QUAL2Kw were run for scenarios with lowered SOD. These results indicate that if SOD levels of post-dredged sites were reduced by removal of loose, organic-rich sediments in addition to vegetation removal DO will rise correspondingly. A 10% reduction in SOD resulted in a 10% increase in DO (see Figure 6-29). An even greater reduction of SOD by 50% increased DO by approximately 90% (Figure 6-30).

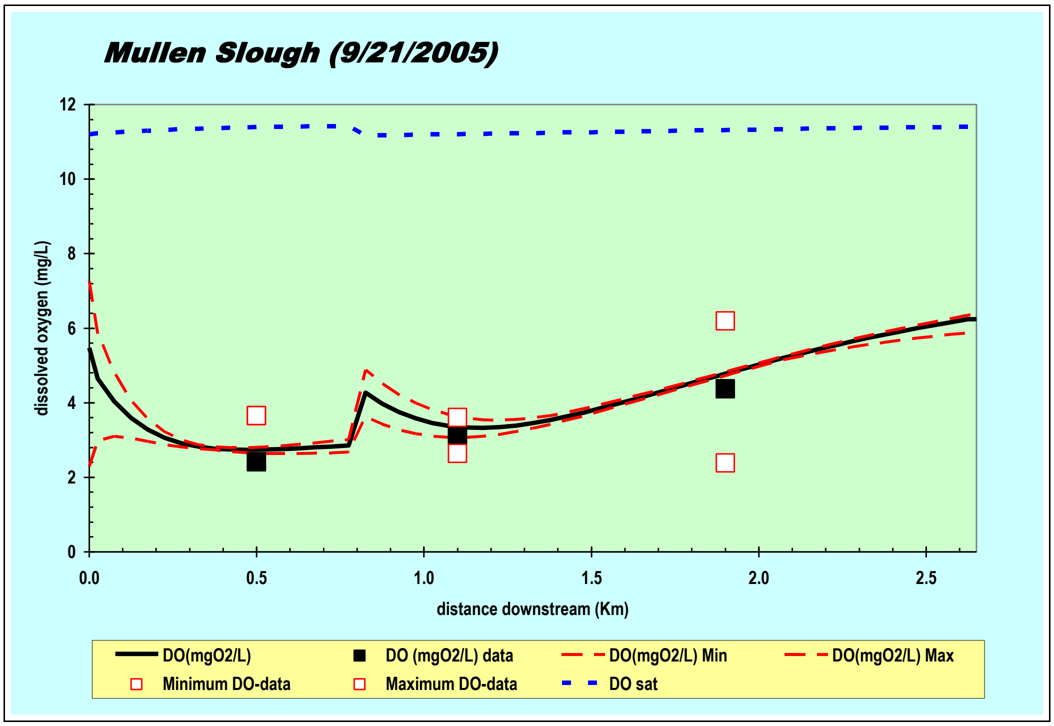

Figure 6-29. QUAL2Kw post-dredge results for a 10% reduction in SOD over current hand-cleaned levels.

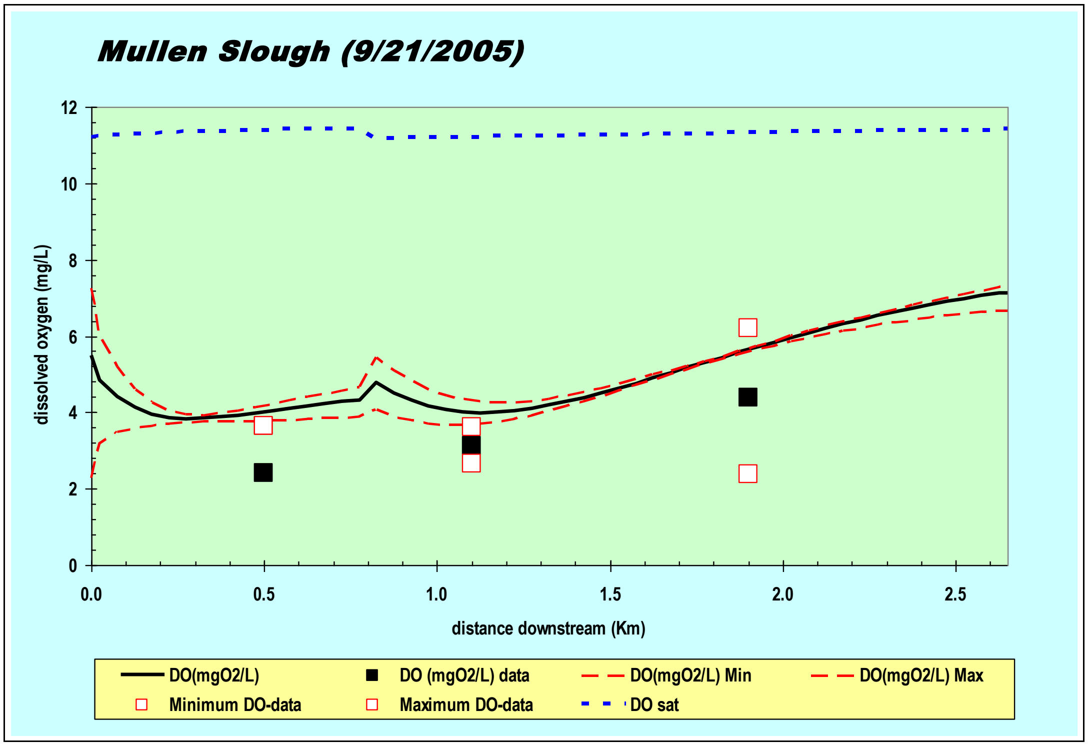

Figure 6-30. QUAL2Kw post-dredge results for a 50% reduction in SOD over current hand-cleaned levels.

6.6.4 DO Results

Both monitoring and modeling indicate that DO levels rise significantly following vegetation removal in agricultural waterways. This increase was observed both at the Mullen Slough and Boscolo sites where DO increased on average (3.4 mg/l) post-dredged. QUAL2Kw predicted an increase of approximately 4.0 mg/l due to vegetation removal. This demonstrates that dredging should potentially be beneficial to the water quality of the systems in question. According to the DOE DO levels must be between 6.5 and 9.5 mg/L for most fish species to survive (Appendix B). Immediately after dredging the oxygen levels were still not quite reaching acceptable levels, which leaves room for additional improvement. This deficit could be due to the amount of resuspended sediment in the channel following dredging as a result of the hand removal technique. Numerous studies have documented the contribution of resuspension to the SOD/BOD sink term in the DO equation (Matlock et al. 2003, Litton 2003).

Both SOD and reaeration rates impacted modeled DO values. QUAL2K calibration resulted in pre-dredge reaeration rates of approximately 1 /day and post-dredged rates of 8.5 /day. This indicates that vegetation removal facilitates increased oxygenation through the air-water interface by removing hindrances to flow and increasing surface area exposed to wind. While vegetation removal enhanced overall water quality the SOD levels in dredged channels actually increased, becoming more of a DO sink. This could be due to the resuspension mentioned above and could be ameliorated by ensuring complete vegetation removal to decrease detritus and decomposition that have been documented as having an adverse effect on water systems' DO levels (Perna and Burrows 2005).

Other factors potentially contributing to the King County system's DO improvement include reduced residence time, enhanced reaeration, and increased algal photosynthesis through enhanced solar radiation exposure. Residence time is lowered when the velocity of the cleaned channels increases due to reduced obstructions, allowing a faster travel time from upstream to downstream. Decreased residence time limits the opportunity of BOD-generating processes to consume oxygen within a reach segment. This is supported by Thomann and Mueller (1987) who express the location of the critical maximum DO deficit (X_c) as a function of the stream velocity multiplied by the critical time of travel. As a result, increased velocity moves X_c downstream because less oxygen is consumed upstream.

A more likely scenario is that reduced residence time contributes to better mixing especially where non-agricultural inflow enters the reach. Vegetation removal allows the incoming flow to reach the main channel more rapidly increasing its ability to introduce healthier oxygen levels. Reaeration is improved when the water surface is exposed to wind after thick vegetation has been removed. An enhanced exposure to rainfall has also been proven to increase reaeration rates (Belanger and Korzum 1991) which prior to cleaning would have been restricted by the plant's interference to precipitation directly entering the water surface. The removal of plant stalks also facilitates wildlife mobility in the channels as noted by WSU researchers who observed renewed waterfowl activity immediately following and even during cleaning events. The reintroduction of animals in the channel contributes to reaeration through their swimming actions which agitate the water surface.

It should be pointed out that both SOD and DO results could be locally impacted by hyporheic flow conditions in several watercourses. Hyporheic (interstitial) flow represents the subsurface flow percolating through the streambed sediments under and beside the open streambed. The source of this flow may be the stream itself or it may come from groundwater entering the stream from the surrounding drainage area. In Oregon, Fernald et al. (2006) found significant surface water quality changes could be linked to hyporheic flow path processes and water exchanges across the streambed boundary. Similarly, Boulton et al. (1998) had earlier reported that exchanges of water, nutrients, and organic matter occurred in response to upwelling and downwelling along gradients that varied spatially and temporally. Although determining regional groundwater movement and hyporheic exchanges was beyond the scope of this study, it should be noted that several, if not all, the agricultural water courses in King County could be potentially impacted (positively or negatively) by these exchange processes thus impacting results.

6.6.5 Final Recommendation

In conclusion, Hypothesis 1 was disproved by monitoring efforts which indicated with certainty that oxygen levels in agricultural waterways differ significantly from those in adjacent non-agricultural reaches. Hypothesis 2 was also rendered null by monitoring and modeling results showing maintenance activities have been demonstrated to improve dissolved oxygen levels in the water column. However, high SOD as a result in remaining dredge spoils hampers further improvement of DO up to EPA standards. The length of this increased SOD was not investigated in this study so it is impossible to pronounce whether the SOD increase is temporary or a permanent result of maintenance activities. In order to augment the improvement to DO levels due to vegetation removal, a process for removing the loose sediment must be employed as well. If more mature soils (soils that had been in place longer) were reached and left exposed to the overlying flow SOD values similar to the undredged sites would theoretically bring DO levels closer to accepted standards.

It is this report's recommendation to continue with hand-removal of vegetation but to consider mechanical dredging to increase sustainable habitat and elevate dissolved oxygen concentrations to even higher levels. While the downstream movement of sediment within watercourse systems may reduce the benefit of mechanical dredging, the overall net benefit of removing sediment from the agricultural waterways is undisputable. Although hand dredging may take longer than mechanical dredging, the simplification in permitting may result in faster total project times. In addition to the significant improvement in dissolved oxygen, there was anecdotal evidence of improved upstream fish passage in the Mullen Slough/Boscolo area that supports the goal of trying to increase usable fish habitat in agricultural waterways.

6.7 References

Belanger, T.V. and Korzum, E.A. 1991. Critique of floating-dome technique for estimating reaeration rates. Journal of Environmental Engineering, 117, pp. 144–150.

Berge, H., Mattila, J., and Carrasco, K. 2000. 1999 Annual monitoring report. King County Agricultural Drainage Assistance Program. Department of Natural Resources and Parks, Seattle, WA. ftp://dnr.metrokc.gov/dnr/library/2002/kcr754.pdf

Berge, H. 2002. 2001 Annual monitoring report. King County Agricultural Drainage Assistance Program. Department of Natural Resources and Parks, Seattle, WA. ftp://dnr.metrokc.gov/dnr/library/2002/kcr763.pdf

Beutel, M. 2003. Hypolimnetic anoxia and sediment oxygen demand in California drinking water reservoirs. Lake and Reservoir Management, 19(3), 208-221.

Beutel, M. 2006. Lake Perris sediment-water interface study: Draft report. Washington State University, Pullman, WA.

Beutel, M., Burley, N., and Culmer, K. 2006. Quantifying the effects of water velocity and oxygen concentration on sediment oxygen demand. Washington State University, Pullman, WA.

Boulton, A.J., Findlay, S., Marmonier, P., Stanley, E.H. and Valett, H.M. 1998. The Functional Significance of the Hyporheic Zone in Streams and Rivers. Annual Review of Ecology and Systematics, Vol. 29, pp. 59-81.

Bowie, G.L. 1985. Rates, constants, and kinetics formulations in surface water quality modeling. Rep. No. EPA/600/3-85/040, Environmental Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Athens, GA.

Brown, L.C. and Barnwell, T.O. 1987. The Enhanced Stream Water Quality Models QUAL2E and QUAL2E-UNCAS: Documentation and User Manual. EPA/600/3-87/007, Athens, GA.

Ecology. 2006. Fresh water designated uses and criteria. WAC 173-201A-200. Olympia, WA. http://apps.leg.wa.gov/WAC/default.aspx?cite=173-201A-200.

Fernald, A.G., Landers, D.H., and Wigington, P.J. 2006. Water quality changes in hyporheic flow paths between a large gravel bed river and off-channel alcoves in Oregon, USA. River Research and Application, Vol 22, No. 10, pp 1111-1124.

Litton, G.M. 2003. Deposition rates and oxygen demands in the Stockton deep water channel of the San Joaquin River. June – November 2001. Prepared for San Joaquin River Dissolved Oxygen TMDL Steering Committee and CALFED. Stockton, CA.

Matlock, M.D., Kasprzak, K.R., and Osborn, G.S. 2003. "Sediment Oxygen Demand in the Arroyo Colorado River." Journal of the American Water Resources Association, 39(2), 267-274.

Nakamura, Y. and Heinz, G.S. 1994. Effect of flow velocity on sediment oxygen demand: Theory. Journal of Environmental Engineering, 120(5), 996-1016.

O'Connor, D.J. and Dobbins, W.E. 1958. Mechanism of reaeration in natural stream. ASCE Transactions, Paper No. 2934, pp. 641–684.

Pelletier, G.J., Chapra, S.C., and Tao, H. 2005. QUAL2Kw - A framework for modelling water quality in streams and rivers using a genetic algorithm for calibration. Environmental Modelling and Software, 21, 419-425.

Pelletier, G.J. and Chapra, S.C. 2006. QUAL2Kw theory and documentation (version 5.1), A modeling framework for simulating river and stream water quality. Washington State Department of Ecology. Olympia, WA. http://www.ecy.wa.gov/programs/eap/models/.

Perna, C. and Burrows, D. 2005. "Improved dissolved oxygen status following removal of exotic weed mats in important fish lagoons of the tropical Burdekin River Floodplain, Australia." Marine Pollution Bulletin, 51, 138-148.

Truax, D.D., Shindala, A., and Sartain, H. 1995. Comparison of two sediment oxygen demand measurement techniques. Journal of Environmental Engineering 121[9], 619-624.

U.S. ACE. 2002. HEC-RAS river analysis system: application guide. U.S. Army Corps of Engineers, Davis, CA. http://www.hec.usace.army.mil/software/hec-ras/

U.S. EPA. 1986. Ambient water quality criteria for dissolved oxygen. U.S. Environmental Protection Agency. Office of Water Regulations and Standards. Criteria and Standards Division. EPA 440/5-86-003. Washington, D.C.

Walker, R. R., and Snodgrass, W. J. 1986. Model for sediment oxygen demand in lakes. Journal of Environmental Engineering 112[1], 25-43.

Warren, C.E., Doudoroff, P., and Shumway, D.L. 1974. Development of dissolved oxygen criteria for freshwater fish. US Environmental Protection Agency, Office of Research and Development, EPA-R3-73-019, Washington, D.C.

Wetzel, R.G. 1983. Limnology, 2nd edition. Saunders College Publishing, Philadelphia, PA. 860 pp.

Appendix 6-A: SOD Sampling Site Maps

Figure A-1. Smith Brother site with Mullen Slough sampling points

Figure A- 2. Boscolo property with sampling points

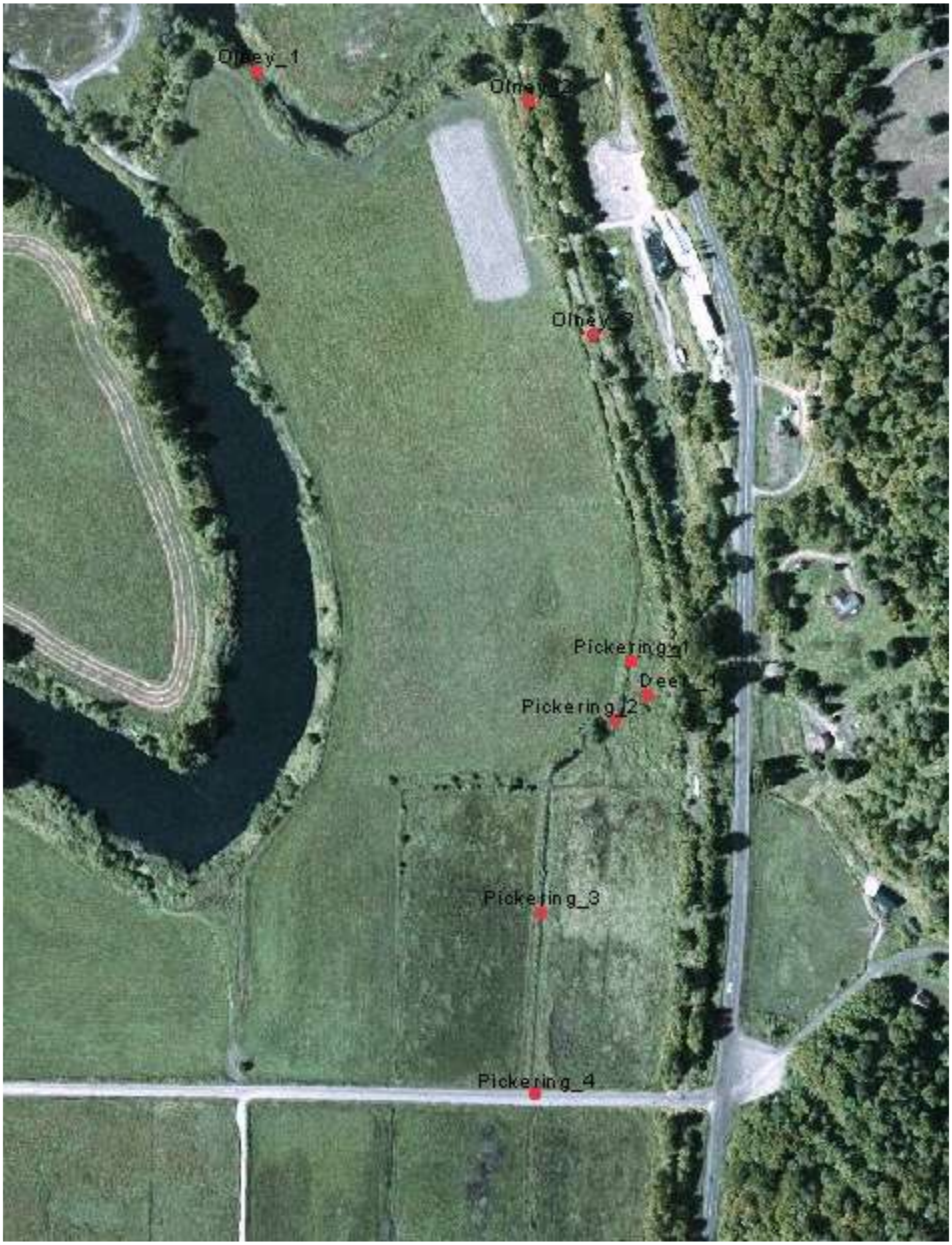

Figure A- 3. Duvall site with Pickering/Olney sampling points

Appendix 6-B: DOE Water Quality Standards

Table B-1 DOE Water Quality Standards*

Category	**Lowest 1-Day Minimum**
Char Spawning and Rearing	9.5 mg/L
Core Summer Salmonid Habitat	9.5 mg/L
Salmonid Spawning, Rearing, and Migration	8.0 mg/L
Salmonid Rearing and Migration Only	6.5 mg/L
Non-anadromous Interior Redband Trout	8.0 mg/L
Indigenous Warm Water Species	6.5 mg/L

*Table can be found in "Water Quality Standards for Surface Waters of the State of Washington" Chapter 173-201A WAC, Table 200(1)(d), page 12
http://www.ecy.wa.gov/pubs/0610091.pdf